

# Temporal changes in south east Tasmanian saltmarshes



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## Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any tertiary institution, and to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

Signed:



Vishnu N. Prahalad  
24<sup>th</sup> August 2009

## Abstract

*Coastal saltmarshes are unique and highly productive ecosystems. They form a major part of the enclosed waterways of south east Tasmania, especially within the Pitt Water, Pipe Clay Lagoon and Ralphs Bay areas. The Pitt Water saltmarshes are acknowledged to be the most diverse and extensive of Tasmanian saltmarshes with extremely high floristic and faunal values, some recognised internationally while some remain unstudied. A study conducted in 1975 detailed the extent and vegetation community composition of the saltmarshes in the Pitt Water, Pipe Clay Lagoon and Ralphs Bay areas. More than 30 years since then, the change in these important ecological communities remained unstudied. Elsewhere in Australia and internationally, temporal studies of saltmarshes have reported substantial changes in morphology, extent and vegetation. This has led to several policy and management actions directed at conserving the function and values of saltmarshes. The objective of the present study was to investigate the spatial and temporal changes in saltmarshes mapped in 1975 to inform policy and planning concerning their management and conservation.*

*Most saltmarshes have had losses in their areal extent, with nearly 17 hectares of area (5%) lost on the seaward side primarily due to coastal erosion. The saltmarsh shorelines have been eroding at about 6 cm to 20 cm a year, with erosion highest on the shorelines more exposed to wind-generated waves. Nearly 6% of the saltmarsh area had been lost due to land reclamation. While some area had been gained through accretion and landward transgression, it was less than a quarter of the saltmarsh area lost. Results for vegetation change show that low marsh plants, which are more adapted to waterlogging, have replaced long lived high marsh plants as the dominant vegetation community of the saltmarshes. A general shift of vegetation zones inland was observed, suggesting a response to sea level rise. Extensive areas within the marsh have been denuded of plant cover and have turned to salt pans/flats and “rotten spots” possibly affected by an increase in tidal inundation, increases in soil salinity related to climate change and increased nutrient inputs from irrigated land. These changes have several implications for the conservation of both the floristic and faunal values of saltmarshes and their contribution to the health and productivity of the coastal waterways. This study highlights a compelling need for strategic planning for the future conservation and management of these important coastal ecosystems in a time of change and contributes key information and methods to assist these efforts.*

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# Chapter 1: Introduction

## 1.1 Research Background

### 1.1.1 Saltmarshes – Values and Threats

“... salt marshes contributed to estuarine food chains beyond their borders and had a greater ecological (and economic) value beyond just being there as open space.”

– Teal and Howes (2000, p. 10)

Saltmarshes are dynamic and productive ecosystems which harbour a wide range of highly specialised flora and fauna species. They provide a range of ecosystem services which are largely determined by their “location, size, and relationships with adjacent land and sea areas” (Doody, 2008, p. 57). These services include, *inter alia*: protecting the hinterland against coastal surges; providing a source of organic materials required for estuarine and marine food webs (supporting coastal fisheries); filtering out contaminants (e.g. nitrates) in freshwater runoff from land before it reaches the sea; conserving biodiversity (e.g. critical habitat for migratory birds); and providing many educational and recreational opportunities (Boorman, 1999; Deegan *et al.*, 2000; Teal and Howes, 2000; Doody, 2008).

In many parts of the world they play a significant role in the nutrient dynamics of adjacent mangroves and seagrass meadows (Bridgewater and Cresswell, 1999; Valiela and Cole, 2002). For this reason, saltmarshes are referred to as being “units within coastal mosaics” that provide several valuable services to the land/seascape they partly constitute (Valiela *et al.*, 2000). In a valuation of ecosystem services provided by each biome, Costanza *et al.* (1997) estimated that estuaries provide the highest value per hectare compared to all other ecosystem types. It is well known that saltmarshes form an important component of estuaries, especially in the temperate regions of the world (e.g. in Australia, Bucher and Saenger, 1991), and play a critical role in estuarine nutrient cycling (Merrill and Cornwell, 2000).

A lack of understanding and appreciation of the services provided by saltmarshes have led to their inappropriate management and consequent decline. Saltmarshes have been extensively grazed, burned, trampled (by humans, domesticated hard-hoofed animals

and vehicles), filled (for land reclamation and waste dumping), restricted by altering hydrological regimes, and subjected to eutrophication (Laegdsgaard *et al.*, 2009). Furthermore, in recent decades, climate change and the related sea-level rise have acted as a severe threat to many coastal saltmarshes (e.g. Boorman, 1992; Adam, 2002).

### **1.1.2 The Saltmarsh Ecosystem**

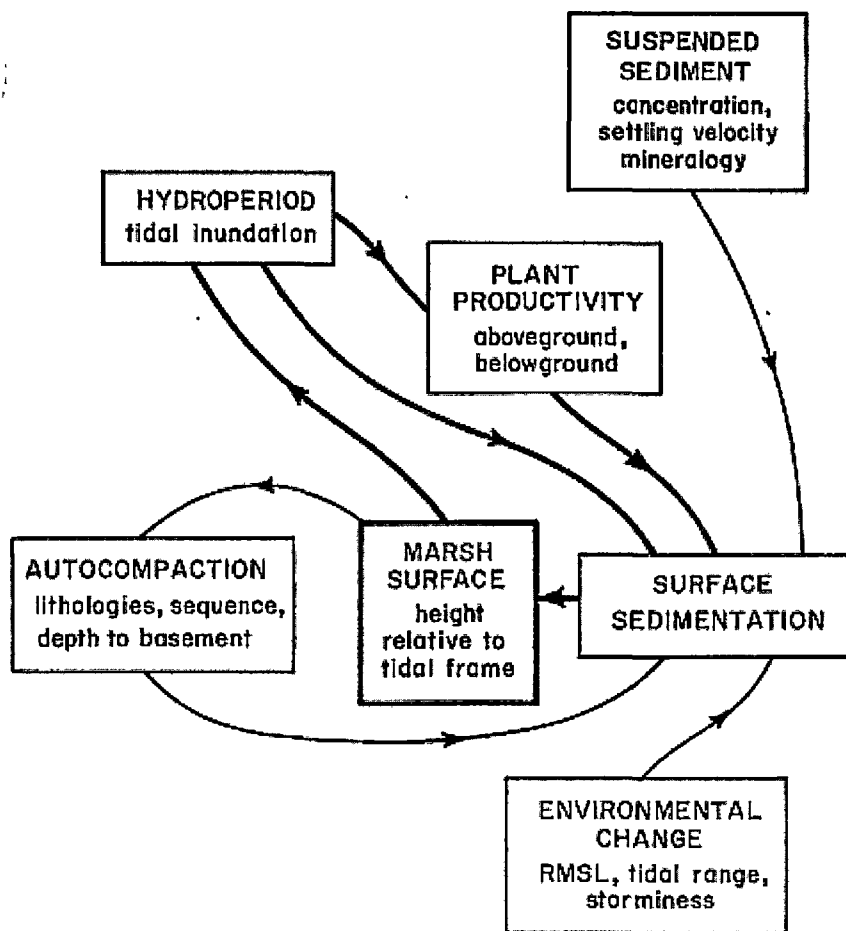
Saltmarshes are vegetation dominated by halophytic plants that are subject to inundation (Chapman, 1974; Adam, 1990). They occur widely in the wetlands of arid and semiarid zones as well as on the coast where environmental conditions, mainly salt and fine sediment input, favour their establishment (Plate 1.1; Plate 1.2). For the purposes of this thesis, coastal saltmarshes (henceforth called saltmarshes) are defined as:

*tracts of land tidally connected to the sea and covered with phanerogamic halophytic vegetation comprised of herbs, shrubs, grasses, sedges and rushes, and including the associated tidal channels, salt flats and marsh pools.*

Vegetation plays a central role in the saltmarsh ecosystem and “provides the environmental structure occupied by the fauna” (Adam, 1990, p. 58). The position of each type of saltmarsh vegetation within the marsh depends on several environmental factors, with salinity and waterlogging regarded as the most consequential (e.g. Clarke and Hannon, 1971; Kirkpatrick and Glasby, 1981). Salinity and waterlogging are predominantly controlled by tidal regimes. Hence tidal influence is considered by many to be the single most important process in the development, extent and function of the saltmarsh ecosystem (e.g. Chapman, 1974; Huiskes, 1990).

### **1.1.3 Temporal Changes in Saltmarshes**

The two temporal changes in saltmarshes of particular concern here are change in saltmarsh ecosystem extent as a whole and change in plant communities within the marshes. These changes result when one or more of the environmental factors that influence saltmarshes change. Both changes to saltmarsh morphology (which affects the extent) and vegetation are illustrated by conceptual linkage diagrams obtained from Allen (2000; Figure 1.1) and Clarke and Hannon (1969; Figure 1.2) respectively.



**Figure 1.1.** Linkages between different “forcing factors” that determine the morphodynamics of the saltmarsh ecosystem (from Allen, 2000, p. 1161). RMSL - relative mean sea level.

Allen (2000) describes four main “forcing factors” that shape the “complex morphosedimentary systems” that constitute saltmarsh.

1. Environmental Change – including changes to sea levels, tidal range and storminess.
2. Sediment Supply – mineral sediment supply, available to the marsh (allochthonous sediment supply).
3. Plant Productivity – both above and belowground plant primary productivity (autochthonous sediment supply and sequestration of allochthonous sediments).
4. Autocompaction – the lowering of the marsh due to autocompaction of the sediment.

These four factors act together to determine the surface sedimentation rates of the marsh, which in turn maintain the marsh surface at a height relative to the tidal frame

(marsh elevation). If one or more of the above mentioned forcing factors change, it will be accompanied by a commensurate change in surface sedimentation and marsh elevation, which then feeds back through changes in hydroperiod and autocompaction causing a morphological change in the saltmarsh. In the case of an accelerated rise in the relative sea level, if the sedimentation (both autochthonous and allochthonous) is not able to maintain the marsh surface relative to the tidal frame, marsh edge erosion or even drowning might result (Orson *et al.*, 1985; Schwimmer and Pizzuto, 2000). Where low lying uplands exist, saltmarshes may be able to migrate inland.

Clarke and Hannon (1969) have provided the linkages that are considered to be consequential in forming the “holocoenotic complex” that determines the extent and nature of plant cover in a given saltmarsh. They suggested that the marsh elevation and slope are the main determinants of the extent of tidal inundation and salt inputs within the marsh, which are further moderated by freshwater sources, water table, rainfall and temperature. These factors combine together to shape the vegetation cover of the saltmarsh. Clarke and Hannon (1971) showed that species varied in their tolerance to salinity and waterlogging. If the tidal inundation pattern is changed with a rise in sea level, changes in salinity and waterlogging will result, which may then be followed by vegetation changes in the saltmarsh (Vanderzee, 1988; Huiskes, 1990).

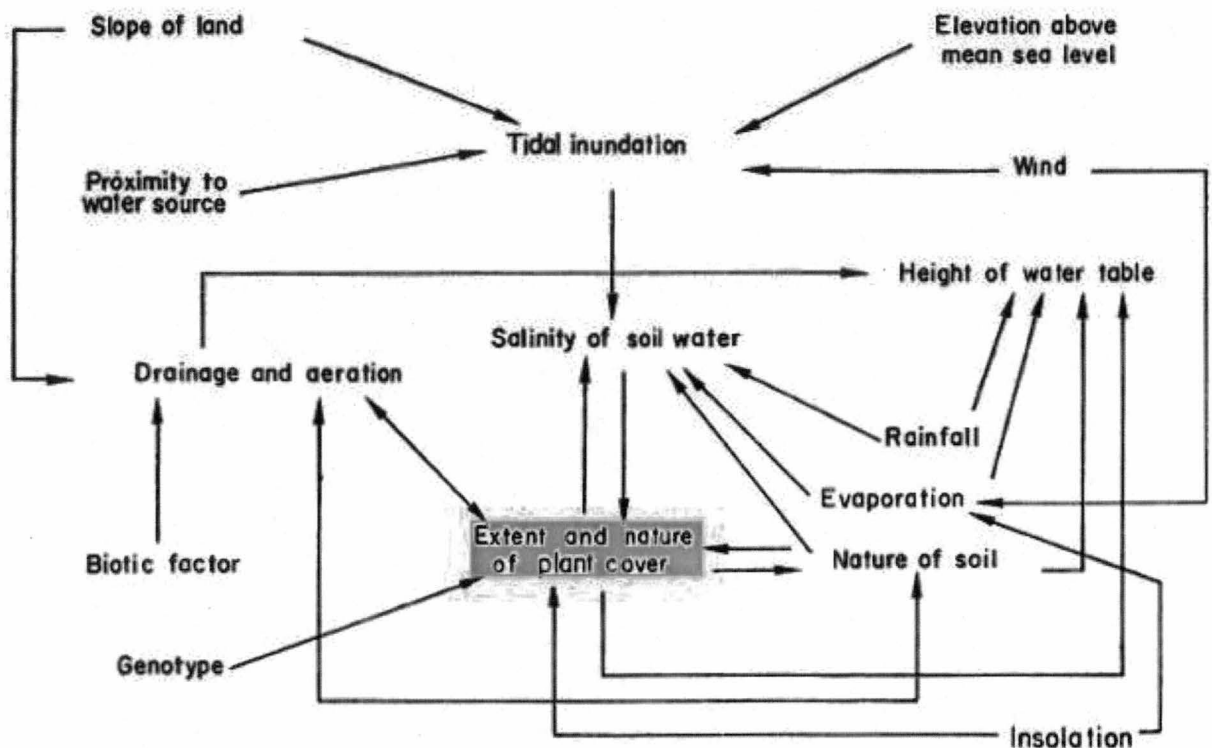
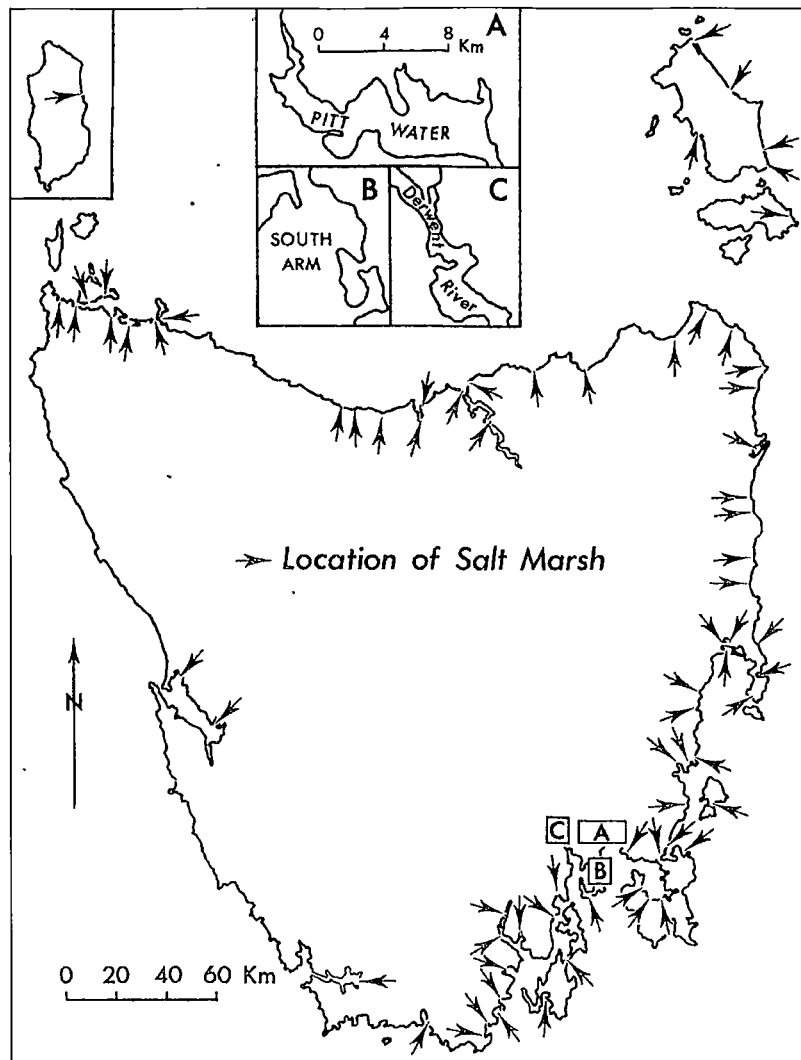


Figure 1.2. The various environmental factors forming the “holocoenotic complex” that is influencing the saltmarsh vegetation communities similar in structure and composition to that found in Tasmania (from Clarke and Hannon, 1969, p. 214; also see Adam, 1990, p. 147).

#### 1.1.4 Saltmarshes in Tasmania and the Research Gap

The most comprehensive study of the vegetation and distribution of saltmarshes in Tasmania was that of Kirkpatrick and Glasby (1981). They mapped the occurrence of saltmarshes in Tasmania (Figure 1.3) and documented the distribution of 35 saltmarsh species. Their distribution maps indicated that most saltmarshes occurred on the sheltered low rainfall locations of the east and south-east coast, especially within estuaries (e.g. Pitt Water, Derwent; as typified in Plate 1.2) and in lagoons (e.g. Moulting Lagoon, Pipe Clay Lagoon). A major exception is the Boullanger Bay and Robbins Passage area to the far north-west of the State, where a drowned sand plain with its islands and sand bars provided sufficient shelter from swell waves for extensive saltmarshes to form (Kirkpatrick and Harris, 1999).





**Figure 1.3. Distribution of saltmarshes around Tasmanian coast as mapped by Kirkpatrick and Glasby (1981). Regions highlighted – A: Pitt Water; B: South Arm; and C: Derwent River – have detailed maps depicting the dominance patterns of saltmarsh vegetation.**

Kirkpatrick and Glasby (1981) also mapped the dominance patterns in the vegetation of 29 saltmarshes spread across three regions, Pitt Water, South Arm and the Derwent River. Based on the vegetation data collected from these 29 marshes, both a structural and a floristic classification were proposed for Tasmanian saltmarsh communities. The floristic classification was based on repeating patterns of co-occurrence of species. The seven floristic classificatory groups may be peculiar to their study area and could not be easily mapped. However, their structural/dominance classification can be widely applied across Australia (Saintilan, 2009) and are mappable. They include communities dominated by: succulent shrubs; grasses; sedges and rushes; and herbs (Kirkpatrick and Glasby, 1981). In later work on saltmarsh, Gouldthorpe (2000) documented the effects of drainage and grazing on vegetation in the Pitt Water and Pipe Clay Lagoon areas and

Morrison (2006) gauged the effects of elevation and sea level on species distribution within two eastern Tasmanian saltmarshes.

Saltmarshes have changed rapidly in Australia over the past few decades with a substantial reduction in their area caused by a number of human and environmental factors (Adam, 1996; Harty, 2004; Laegdsgaard, 2006). One of the recent well documented losses was caused by the incursion of mangroves into saltmarsh habitat due to anthropogenic influences and sea level rise (Saintilan and Williams, 1999). Straw and Saintilan (2006) noted that many estuaries in south-east Australia have lost 25 to 80% of salt marsh cover to “mangrove transgression” and suggested that the trend will increase with the predicted sea-level rise. While several studies have documented changes to saltmarshes in Australia (summarised in Wilton, 2002), change analysis studies for saltmarshes in Tasmania are lacking. The Tasmanian State of the Environment Report 2003 (Resource Planning and Development Commission, 2003) reported that the condition of saltmarshes in the State are “not well known” and suggested that the only comprehensive study of saltmarsh vegetation composition available in Tasmania (Kirkpatrick and Glasby, 1981) could be used to determine the changes and trends in Tasmanian saltmarshes. Furthermore, the report on Tasmanian Coastal Vulnerability to Climate Change and Sea-Level Rise (Sharples, 2006) identified saltmarsh shorelines as being vulnerable to wave erosion. The extent of saltmarsh shoreline erosion and vulnerability to wind waves in Tasmania has not been studied.

## **1.2 Thesis Aim and Research Objectives**

The main aim of the thesis is to identify changes in the *extent* and the *community composition* of the saltmarshes between 1975, as mapped by Glasby (1975), and the present (i.e. 2009) in key south-eastern areas, in order to inform policy and planning concerning the management and conservation of Tasmanian coastal saltmarshes.

To fulfil the thesis aim, the research objectives are:

1. to determine the changes in landward and seaward extent of saltmarshes;
2. to identify any morphological changes within the marshes;
3. to identify areas where obligate saltmarsh species can migrate landwards in the event of sea level rise;
4. to determine the changes in the vegetation composition of saltmarshes;
5. to identify environmental and anthropogenic factors that may have resulted in the above changes; and,
6. to discuss management and conservation implications of the above results.

## **1.3 Structure of the Thesis**

The second chapter describes the study areas and the methods used in the project. The third chapter addresses the first three research objectives related to saltmarsh extent. The fourth chapter addresses research objectives four and five. The fifth chapter addresses the final research objective.



**Plate 1.1.** A small saltmarsh (with *Austrostipa stipoides* and *Tecticornia arbuscula*) found on a rocky shore (in South Arm Peninsula, south east Tasmania). While their establishment was favoured by salt spray and aeolian sediments, growth and expansion is limited due to space/substrate restrictions and also due to the effect of wind.



**Plate 1.2.** The mouth of Sorell Rivulet (in Pitt Water, south east Tasmania) exhibiting some of the aspects of saltmarsh development. A – An ebb delta was able to be colonised by saltmarsh plants because of the protection provided by the nearby headland from the long shore drift (indicated by the arrow). B and C – Saltmarshes developed on the sheltered areas of the estuary. Also note the marsh pool at B which forms an integral part of the saltmarsh and forms similar functions to the sewage settling ponds built in the headland.



## Chapter 2: Study Area and Methods

### 2.1 Study Area

The project covers two *study regions*: Pitt Water and South Arm, within which there are seven *study areas* (Figure 2.1), and are further divided in to 22 *study sites* (Figure 2.2; Figure 2.3). The first section of this chapter provides detailed description of the location, climate, geomorphology, vegetation, and land tenure and land use of these study areas.

#### 2.1.1 Location

The study areas are located in south eastern Tasmania, Australia, within 20 km radius of its capital city, Hobart. They are located between  $42^{\circ} 44'$  and  $42^{\circ} 60'$  southern latitude and  $147^{\circ} 25'$  and  $147^{\circ} 37'$  eastern longitude.

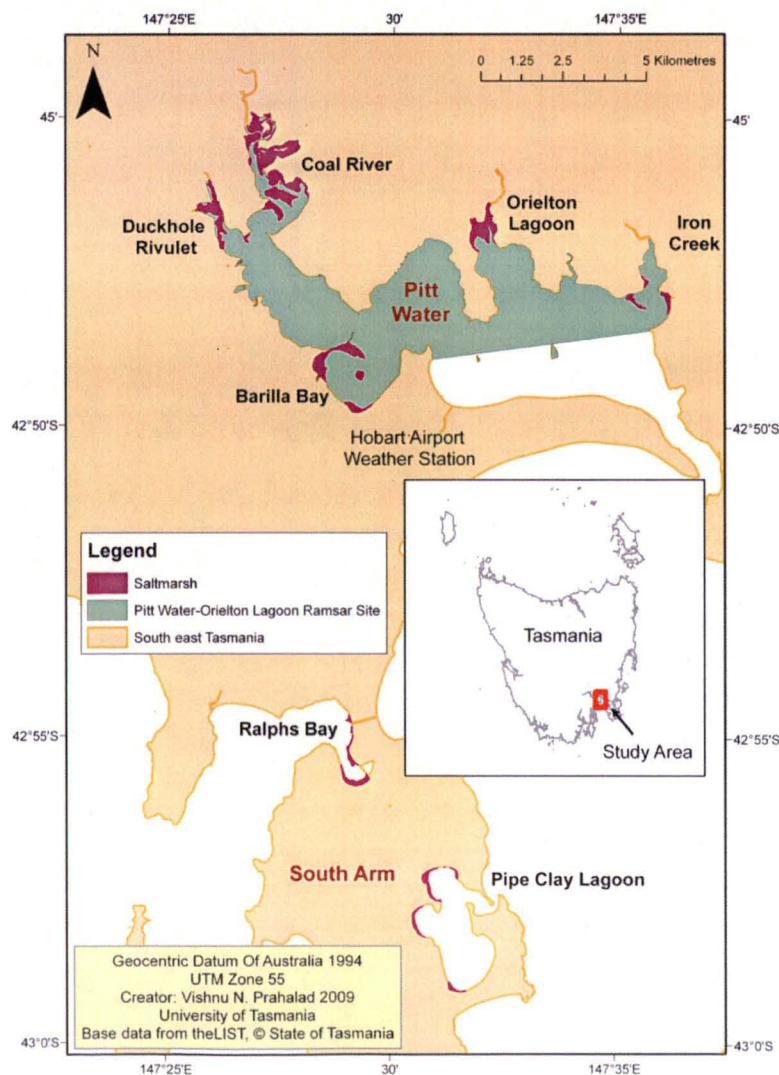


Figure 2.1. Study Areas within the Pitt Water and South Arm regions of south east Tasmania.

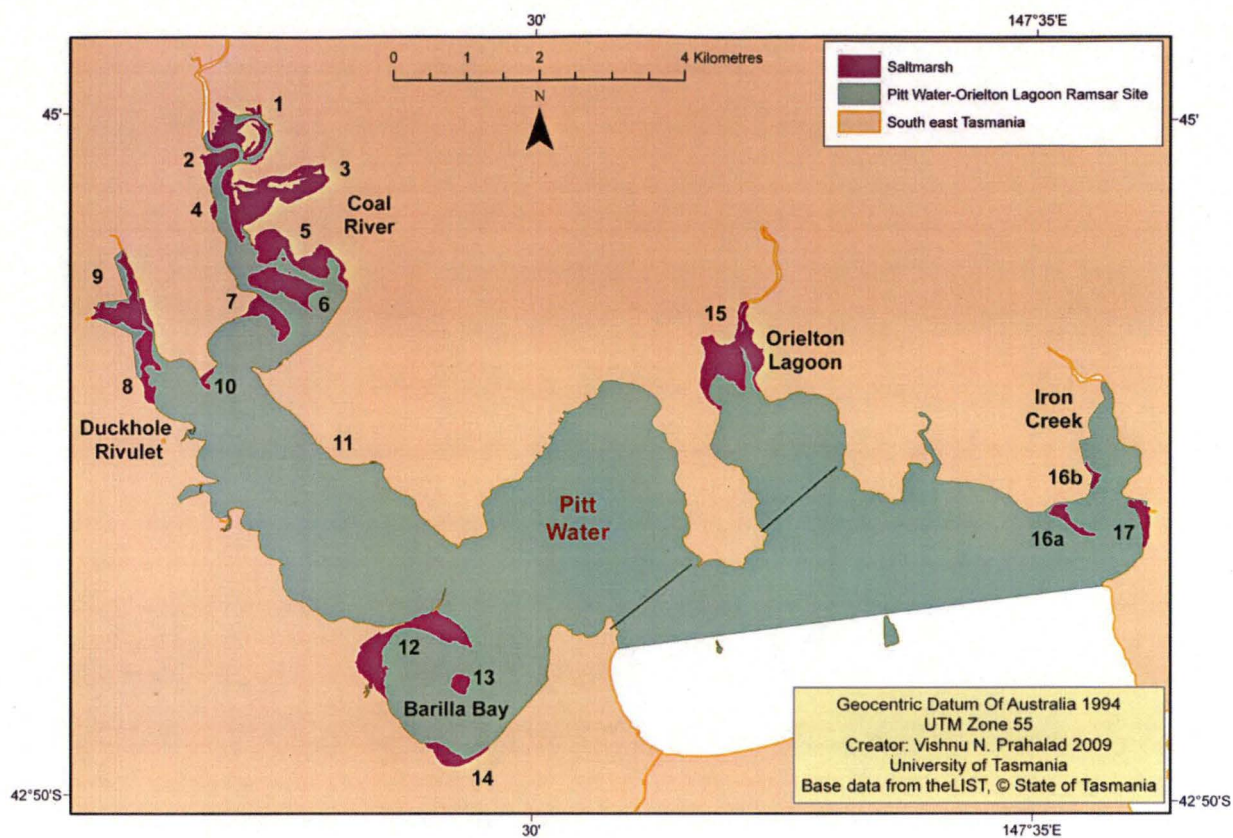


Figure 2.2. Location of study Sites 1 to 17 in the Pitt Water region. Site 11 is not shown to have any saltmarsh as it has been claimed since previous study.

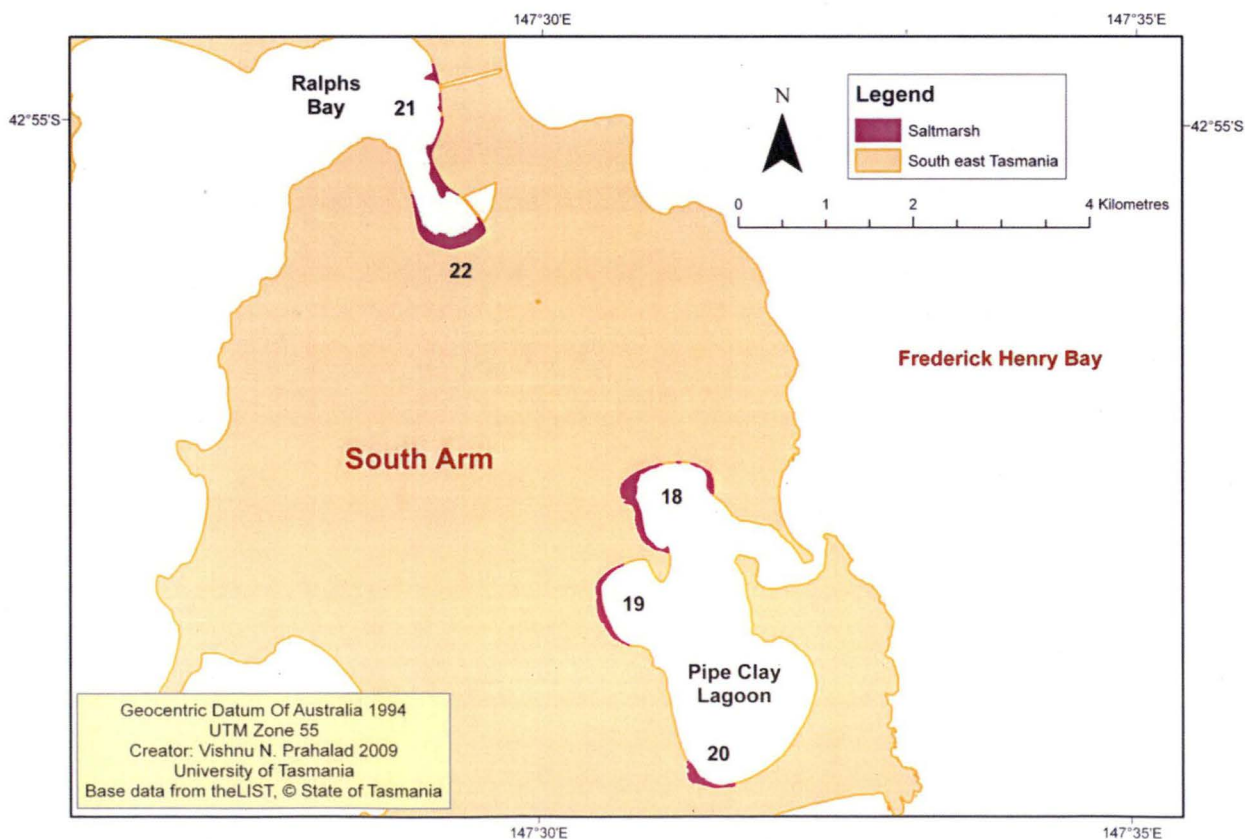


Figure 2.3. Location of study Sites 18 to 22 in the South Arm region.



### **2.1.2 Climate**

The climatic records have been obtained from Hobart Airport (Australian Government Bureau of Meteorology Site No: 094008) located between the Pitt Water and South Arm study area at 42.83° S latitude and 147.50° E longitude (see Figure 2.1). The site is located at an elevation of 4 m and is approximately 10 km to the farthest study site in the Pitt Water area (Site 1) and about 17 km to the farthest in the South Arm area (Site 20).

#### ***2.1.2.1 Rainfall***

The mean monthly rainfall data from Hobart Airport from 1958 to 2008 indicate that rainfall and the mean number of rain days (with rain measuring  $\geq 1$  mm) were almost even across the year with the second half of the year being wetter than the first (Figure 2.4). The mean annual rainfall was 497.1 mm with the highest annual rainfall being 735.4 mm in 1975 and the lowest being 297.2 mm in 2006 (Bureau of Meteorology, 2009).

#### ***2.1.2.2 Temperature***

The mean maximum and minimum temperature recorded at Hobart Airport from 1958 shows that the temperature follows the seasons (Figure 2.5), with highest temperatures in the summer months (December, January, February) and the lowest in winter (June, July, August). January is the hottest month with mean daily maximum and minimum temperatures of 22.5 °C and 12.0 °C. July is the coldest month with mean daily maximum and minimum temperatures of 12.4 °C and 4.1 °C. The mean annual daily maximum temperature was 17.5 °C with the mean annual daily minimum temperature 8.0 °C. The highest temperature recorded at the station was 40.1 °C in January 1991 and the lowest, -3.9 °C in June 1972. The minimum temperature reaches below 2 °C (threshold for a potential frost day) on an average of 21.4 days an year (Bureau of Meteorology, 2009).

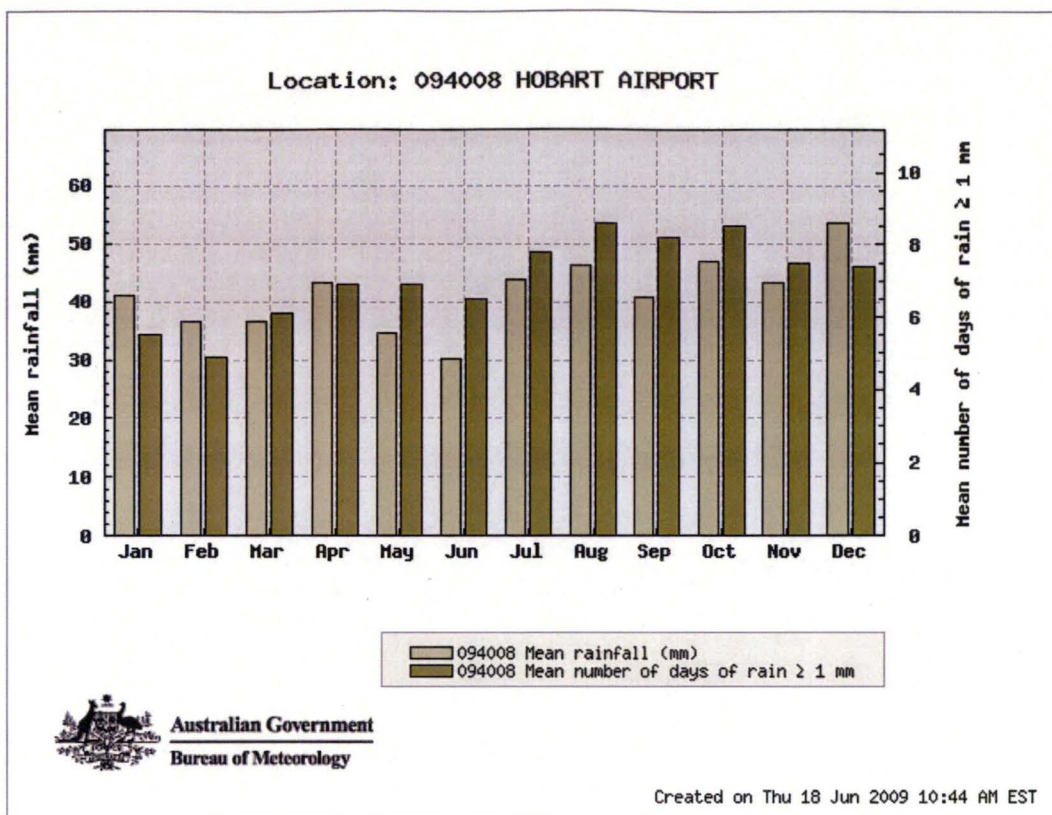


Figure 2.4. Mean annual rainfall (in mm) recorded at Hobart Airport since 1958, shown along with mean number of days of rain  $\geq 1$  mm. (Bureau of Meteorology, 2009)

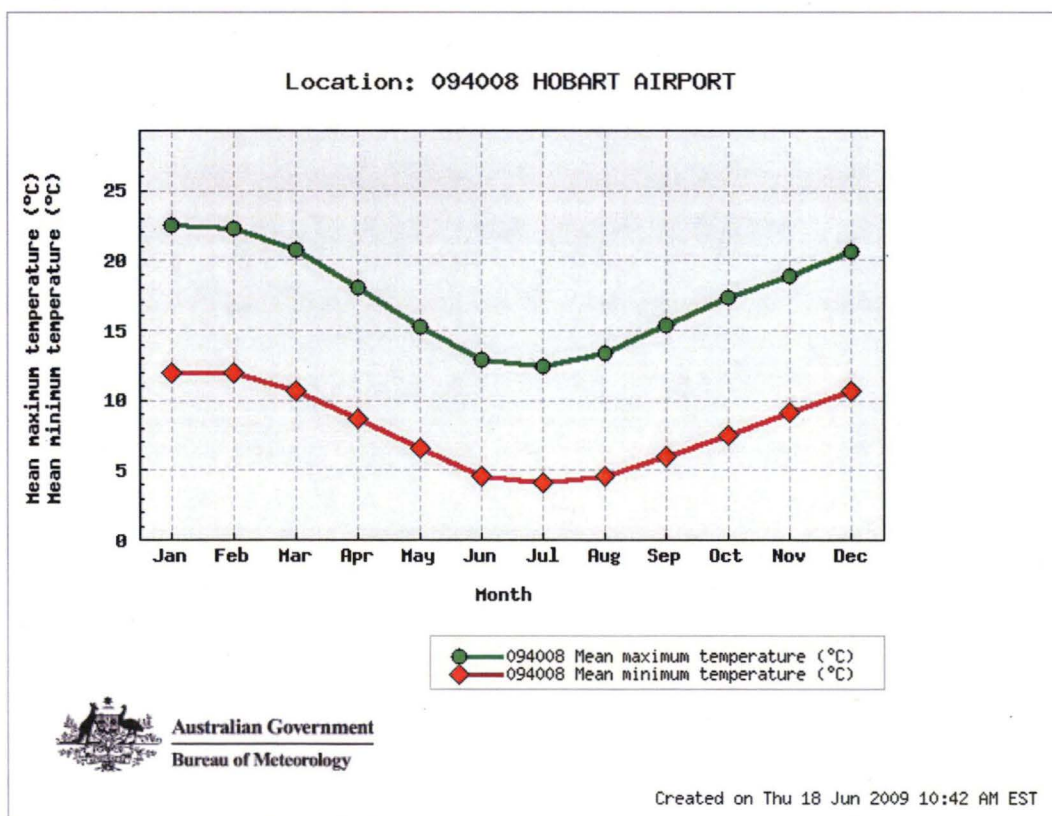


Figure 2.5. Mean maximum and minimum temperature recorded at Hobart Airport 1958-2008. (Bureau of Meteorology, 2009)

### 2.1.2.3 Wind

Wind data recorded at Hobart Airport for 9 am and 3 pm since 1958 indicate that the mean monthly wind speed fluctuates between 12 and 25 km/h with higher mean wind speeds in the warmer months (Figure 2.6). There is a striking positive relationship between the mean monthly 3 pm wind speed and the mean temperature for a given month. Also, the mean 3 pm wind speed is about 10 km/h higher than the mean 9 am wind speed suggesting that, on average, the evenings are windier than the mornings in the warmer months of the year. The wind gust speed records, dating back to 1958, show the maximum wind gust speed to be 130 km/h. The mean daily wind run records were available only from 1995 and the average is 384 km. The mean daily wind run was higher for the warmer months of the year (Bureau of Meteorology, 2009).

Wind rose (wind speed vs direction) diagrams for 9 am and 3 pm show a difference in the general frequency of the morning and afternoon winds (Figure 2.7; Figure 2.8). The 9 am wind direction was predominantly north-westerly while the 3 pm wind direction was mainly south-easterly, while southerly and north-westerly winds were moderately common.

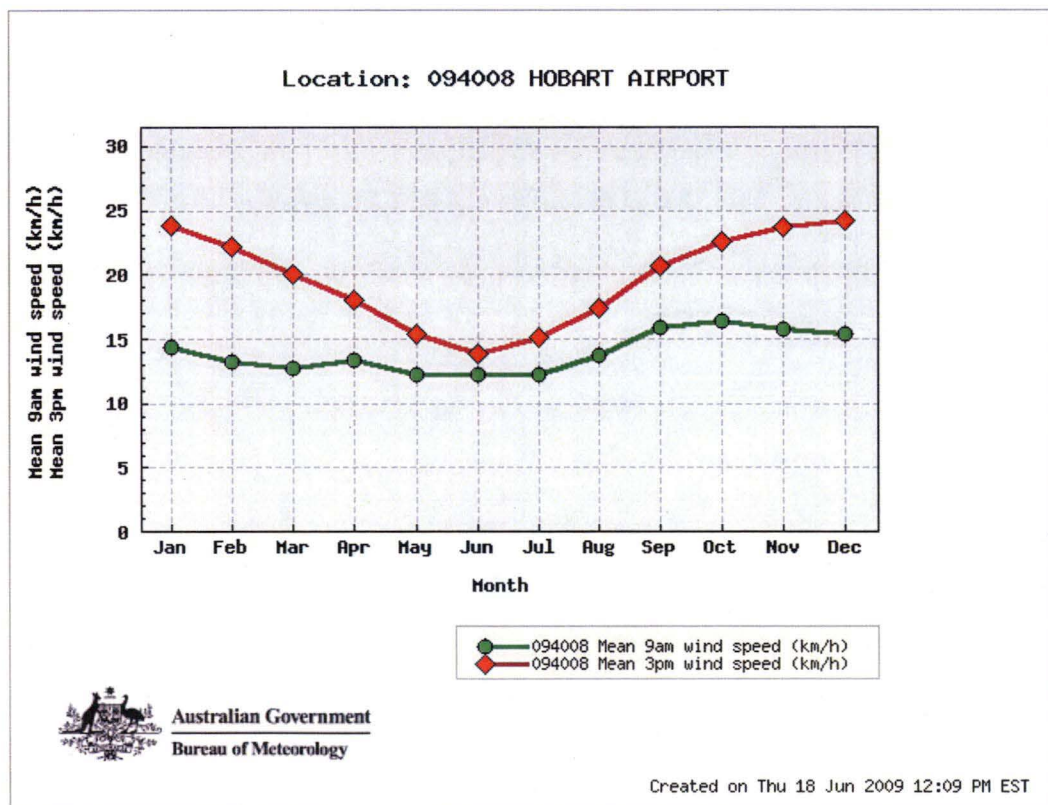


Figure 2.6. Mean monthly wind speed (in km/h) recorded at Hobart Airport since 1958 for 9 am and 3 pm. (Bureau of Meteorology, 2009)



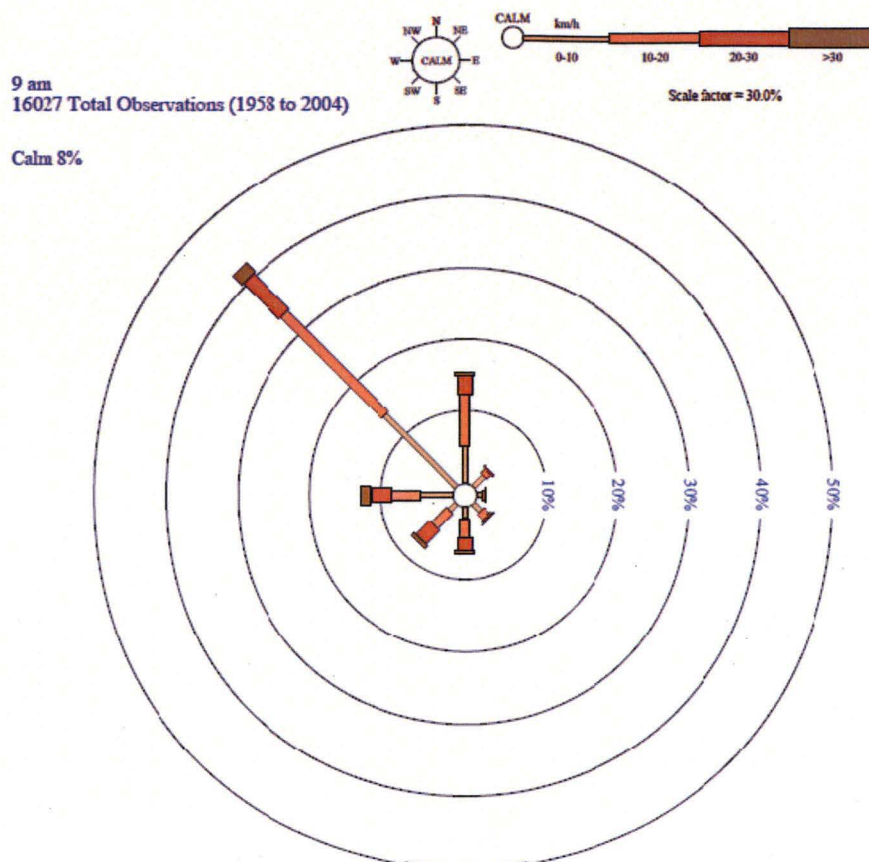


Figure 2.7. Wind Rose diagram obtained from Hobart Airport (1958-2004) showing the wind direction, speed (in km/h) and percentage frequency for eight compass directions for 9 am. (Bureau of Meteorology, 2009)

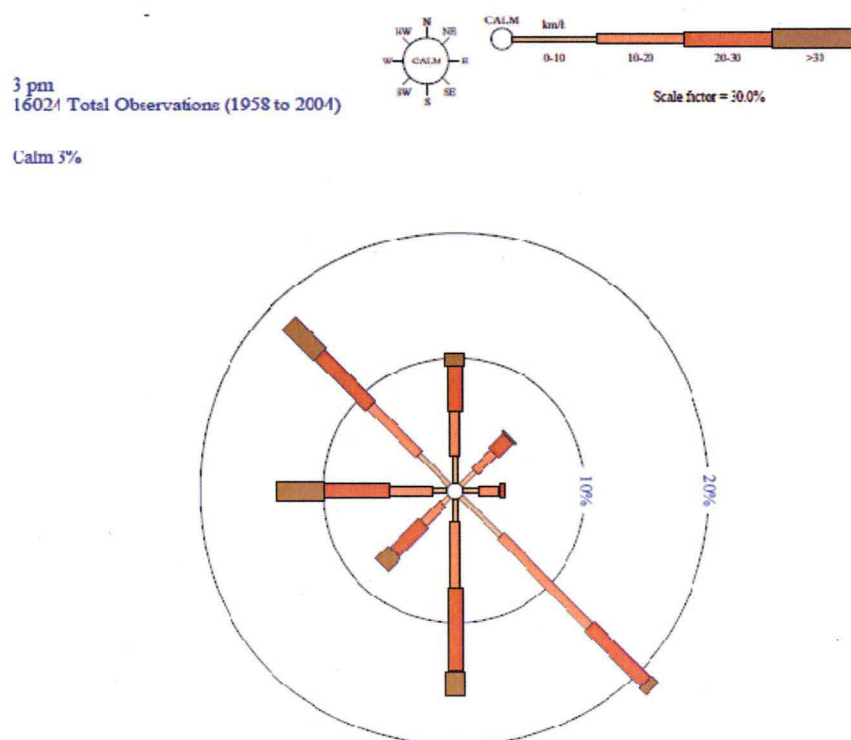


Figure 2.8. Wind Rose diagram obtained from Hobart Airport (1958-2004) showing the wind direction, speed (in km/h) and percentage frequency for eight compass directions for 3 pm. (Bureau of Meteorology, 2009)

### 2.1.3 Substrate

The saltmarshes in the study areas originated during the Holocene and are essentially depositional environments made up of Quaternary sediments (Geological Survey of Tasmania, 1972; 1982; Tasmanian Geological Survey, 2003). The saltmarsh substrates within the Pitt Water region are predominantly made up of alluvial deposits comprising of paralic clay, silt, sand and minor gravel. Substrate composition in Ralphs Bay is predominantly dune and windblown sand, except for the northern most section of Site 21 which constitutes sand and gravel deposited from dredging a canal nearby. Within Pipe Clay Lagoon, Site 19 and the eastern side of Site 18 consists of windblown sand deposits while the other sites consist of river alluvium, swamp, marsh, beach and spit deposits. In general saltmarsh substrates found across the study sites can be broadly grouped under two types: clay/silt and sand/gravel (Figure 2.9; Figure 2.10; Figure 2.11). While some organic content is usually found mixed with the clay/silt substrate types, the Tasmanian saltmarsh substrate cannot be classified as peat (Isbell, 2002).

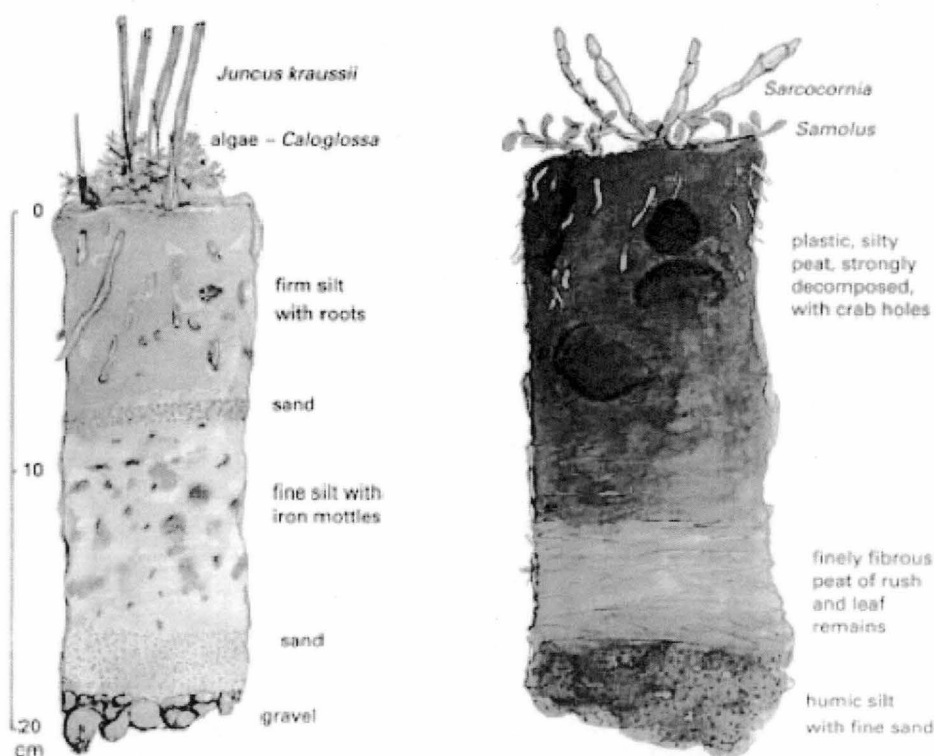


Figure 2.9. Mineral substrate types found in the saltmarsh as depicted by Johnson and Gerbeaux (2004, pp. 142-143) for New Zealand saltmarshes similar to Tasmanian saltmarshes. The *Juncus kraussii*-algae substrate shows layers of gravel, sand and silt (and the depositional history). The grey colour of the silt is due to the gleying of the wet soil in anaerobic conditions. The presence of the iron mottles, which are places of better aeration, is common where water table fluctuates as in the case of a tidal creek mouth in an estuary where the profile illustrated by Johnson and Gerbeaux exists. The *Sarcocornia*-*Samolus* substrate illustrates a clayey, silty, partly organic soil obtained from the mid-tidal zone in an estuarine saltmarsh. The crab holes are means through which mineral sediments are mixed with organic material.





**Figure 2.10. Sand/gravel substrate type with wind/wave deposited shells.**



**Figure 2.11. Clay/silt substrate type with some organic content.**

#### **2.1.4 Vegetation and Marsh Physiography**

Glasby (1975) provided a comprehensive account of the composition and distribution of the saltmarsh vegetation in the study sites. The vegetation was comprised of four main “structural-dominance communities”, dominated by succulent shrubs, grasses, sedges/rushes or herbs (Kirkpatrick and Glasby, 1981). The organisation of these dominance communities within the study sites were largely related to drainage and salinity, with some marshes exhibiting vegetation zonation based on these two environmental gradients. However, the presence and absence of vegetation types within the saltmarsh also depended on the size of the marsh and the effect of localised environmental factors (Glasby, 1975).

An idealised saltmarsh (Figure 2.12) can be divided into the low marsh (LM), middle marsh (MM) and the high marsh (HM) with each section within the marsh having different elevation, inundation frequency and salinity. The change in these physical parameters is usually reflected in the vegetation, with the LM being dominated by herbs and low shrubs, MM being dominated by tall shrubs, and the HM being dominated by sedges, grasses and rushes. A defining aspect of the physiography of the saltmarsh is the tidal channel (TC) network, which functions to distribute tidal water and sediments to the marsh flat/platform (Allen, 2000). Within the study sites, only the sites in Coal River, Barilla Bay and Duckhole Rivulet have well defined creek systems which meander through the marsh, and, in some cases, form a dendritic network (Figure 2.13).

Another important physiographic feature clearly identifiable within the study sites is salt pans or marsh pools depending on the presence or absence of water. These “bare patches” (BP) hold water following heavy rain or spring tides, but remain dry during other times “leaving a surface layer of crystalline salt” (Long and Mason, 1983, p. 24; Figure 2.14). Bare patches form a significant component of the study sites, with some sites having extensive areas of bare ground, mostly in the mid and high marsh. The bare areas generally found in the mid marsh are “pans” forming depressions in the marsh platform and are often poorly drained. The bare areas generally found in the high marsh are gently sloping and is relatively well drained.



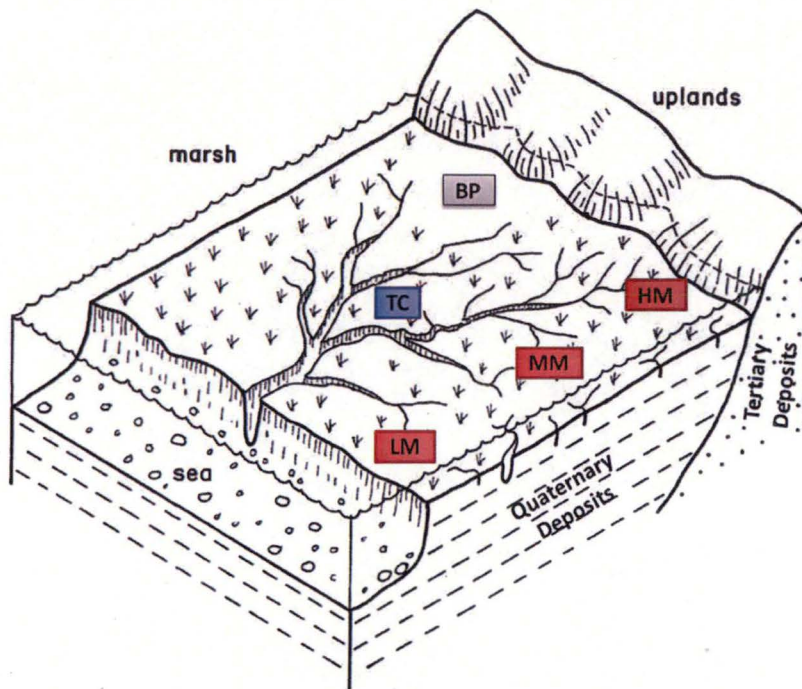


Figure 2.12. A typical saltmarsh (at extreme flood tide) showing the different physiographic features: 1. Marsh platform divided into low marsh (LM), middle marsh (MM) and high marsh (HM); 2. Tidal channels/creeks (TC) cutting into the marsh platform; 3. Un-vegetated bare patches (BP) at the high marsh. Also seen is the marsh scarp/cliff at the seaward edge, and the usually sharp upland preventing landward colonisation of the saltmarsh. Tidal flooding of the marsh is dependent on marsh elevation and the height of the tide. (Illustration adapted from Allen, 2000).



Figure 2.13. Aerial oblique photograph of the sinuous dendritic tidal channels cutting through the mid marsh in Site 12, Barilla Bay.

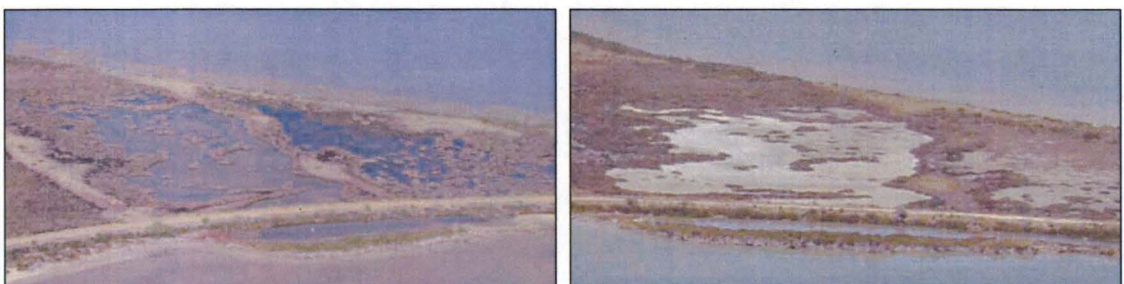


Figure 2.14. Bare patches in the Saltmarshes can either be “salt flats/pans” or “marsh pools” as seen in the two photographs above of the same area (in Site 13, Barilla Bay) taken at two different times. Photo to the right: Hydro Tasmania Consulting.



### **2.1.5 Land Tenure and Land Use**

The Sites 6, 13 and 15 fall entirely within Pitt Water Nature Reserve, the reserved part of the Pitt Water-Orielton Lagoon Ramsar Site (Parks and Wildlife Service, 2009). Sites 8, 9, 10, 16 and parts of 17 fall within the unreserved parts of the Ramsar Site, some on unallocated Crown Land and some on private land. Sites 12 and 2 in Pitt Water have been registered as private reserves by the respective landowners. Both Sites 21 and 22 in Ralphs Bay are Crown Land, some of which is reserved as the Ralphs Bay Conservation Area. Sites 18, 19 and 20 fall partly in Crown Land and partly in private land. Part of Site 20 is in the Pipe Clay Lagoon Conservation Area and parts are within a private reserve voluntarily set up by the landowners.

The land uses on and around the saltmarsh sites vary from nature appreciation and passive recreation to intensive agriculture and dumping fill (Table 2.1). The most prominent land use is grazing by livestock. Large stretches of land behind Sites 5, 8 and 9 are used for cash crops and are actively irrigated (Figure 2.16). Dog walking, horse and bike riding is common in Pipe Clay Lagoon, where the saltmarshes abut residential areas.

**Table 2.1. The area (in m<sup>2</sup>), land tenure and land uses of each study site.**

Region	Study Area	Site No.	Area (in m <sup>2</sup> )	Land Tenure	Land Use
Pitt Water	Coal River	1	184,112	Private Land	Agriculture
		2	99,458	Private Land	Private Reserve
		3	494,701	Private Land	Agriculture
		4	11,348	Private Land	Agriculture
		5	288,939	Private Land	Agriculture
		6	237,085	Nature Reserve	Pitt Water Nature Reserve
		7	120,864	Private Land	Agriculture
	Duckhole Rivulet	8	120,810	Private Land	Agriculture
		9	211,560	Private Land/ Crown Land	Agriculture
		10	14,045	Crown Land	Unallocated Crown Land
	Barilla Bay	12	376,898	Private Land	Private Reserve
		13	43,724	Nature Reserve	Pitt Water Nature Reserve
		14	89,023	Private Land	Cambridge Aerodrome
	Orielton Lagoon	15	430,947	Private Land/ Nature Reserve	Agriculture/ Orielton Lagoon Nature Reserve
	Iron Creek	16a	50,728	Crown Land	Unallocated Crown Land
		16b	13,431	Crown Land	Unallocated Crown Land
		17	51,628	Private Land	Agriculture
South Arm	Pipe Clay Lagoon	18	106,391	Private Land/ Crown Land	Unallocated Crown Land/ Agriculture
		19	47,252	Private Land/ Crown Land	Unallocated Crown Land/
		20	32,871	Private Land/ Crown Land	Private Reserve
	Ralphs Bay	21	55,651	Crown Land	Ralphs Bay Conservation Area
		22	114,828	Crown Land	Ralphs Bay Conservation Area



Figure 2.15. Sheep grazing on *Sarcocornia quinqueflora* in Site 3 in the Coal River area.



Figure 2.16. Irrigated agriculture in the catchment of Site 5 in the Coal River area.  
Photo: I. Houshold.

### 2.1.6 Ecological Significance

The study sites fall within two estuaries, Derwent and Pitt Water, which are considered as high conservation value aquatic ecosystems nationally and listed as the two “priority coastal hotspots” (under Caring for our Country business plan 2009-10) in Tasmania (Commonwealth of Australia, 2009). Pitt Water-Orielton Lagoon area has been noted to have the richest saltmarsh communities in Tasmania (Kirkpatrick and Glasby, 1981). It has some of the most extensive areas of *Tecticornia arbuscula* (saltbush) in Tasmania. The area is home to two saltmarsh plants listed a rare under the Tasmanian *Threatened Species Protection Act 1995*, *Limonium australe* (yellow sea lavender; Figure 2.17) and *Wilsonia humilis* (silky wilsonia).

Several rare and threatened species of fauna occur in the study areas, including the Saltbush Blue Butterfly (*Theclinessthes serpentata*) which occurs in Pitt Water (Figure 2.18) and the Tunbridge Looper Moth (*Chrysolarentia decisaria*) in Ralphs Bay. One of the most important ecological values of the study area that is internationally recognised is its importance for migratory birds as nesting, feeding and roosting habitats. Pitt Water-Orielton Lagoon is one of the ten Ramsar Wetlands of International Importance in Tasmania and provides nesting and roosting space for migratory birds in summer and for resident shorebirds, waterfowl and some terrestrial birds all through the year (Figure 2.19.; Figure 2.20; Parks and Wildlife Service, 2009). Pitt Water-Orielton Lagoon, along with Ralphs Bay and Pipe Clay Lagoon, make up Australia’s southernmost section of the East Asian-Australasian Flyway which stretches up to Siberia and Alaska (Birds Tasmania, 2006).





Figure 2.17. *Limonium australe* (yellow sea lavender) found in Barilla Bay Site 12.



Figure 2.18. Saltbush Blue Butterfly (*Theclinessthes serpentata*) resting on *Austrostipa stipoides* in Barilla Bay Site 12. Also seen in the image is *Tecticornia arbuscula* (Saltbush) in the back and *Atriplex paludosa* to the right.





Figure 2.19. Red-capped Plovers (*Charadrius ruficapillus*) in Site 7 in Coal River.



Figure 2.20. Australian Pelican (*Pelecanus conspicillatus*), Pied Oystercatcher (*Haematopus longirostris*) and Masked Lapwings (*Vanellus miles*) in Site 10 in Coal River.

## 2.2 Methods

### 2.2.1 Data Collection

Through the course of the project, nine different types of data were used. They were:

1. Original vegetation maps created in 1975
2. Aerial photographs from 1965/66 and 2008
3. Latest and best available satellite imagery from 2005
4. Digital Elevation Models (DEM)
5. Vector layers from Land Information Services Tasmania (LIST)
6. Oblique aerial photographs
7. Geographic coordinates from handheld Geographic Positioning System (GPS)
8. Field data and on-field oblique photographs
9. Climatological data from the Bureau of Meteorology (BOM)

#### 2.2.1.1 1975 Vegetation Maps

The vegetation mapping done by Glasby (1975) covered marshes in Pitt Water sites, South Arm and the Derwent Estuary (Kirkpatrick and Glasby, 1981). The original maps which were on distortion-proof transparent paper (Figure 2.21) were held by Professor Jamie Kirkpatrick, who supervised the work done by Glasby in 1975. The ten maps covering the study sites selected for this project were obtained and scanned in a Xerox Synergix Wide Format Scanner YWC-1.

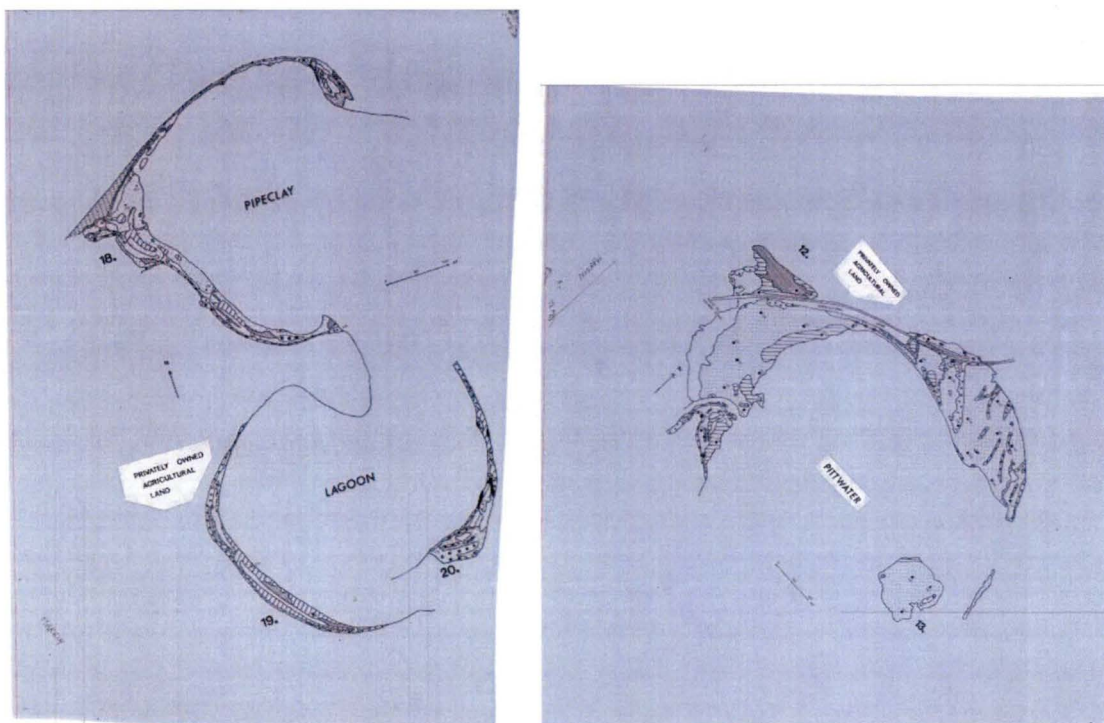


Figure 2.21. Scanned 1975 maps detailing the extent and composition of the saltmarshes.



#### ***2.2.1.2 Aerial Photographs (1965/66 and 2008)***

Two sets of aerial photographs were obtained for the project. Six black and white aerial photographs, those used by Glasby in his mapping in 1975, were obtained from the Geography Resource Centre, School of Geography and Environmental Studies, University of Tasmania (original source: Lands Department, Tasmania, 1965). The photographs for Coal River, Duckhole Rivulet, Barilla Bay and Ralphs Bay areas were taken on 2<sup>nd</sup> February 1965 at low tide, at a flying height of 3,962.4 m and at a scale of 1:31,680. The photographs for Orielson Lagoon, Iron Creek and Pipe Clay Lagoon were taken on 8<sup>th</sup> February 1966 at low tide, at the same height and scale as the 1965 photos. Photographs were scanned in an Epson Expression® 1640XL flatbed scanner as 8-bit greyscale TIFF images at a resolution of 1,200 dots per inch.

The latest available colour aerial photographs produced by TASMAR, Tasmanian Department of Primary Industries and Water, were obtained for the study sites. They included eight photographs taken on 3<sup>rd</sup> January 2008 at low tide, at a flying height of around 3,800 m and at a scale of 1:24,000. They were scanned in an Epson Expression® 1640XL flatbed scanner as 24-bit colour TIFF images at a resolution of 1,200 dots per inch.

#### ***2.2.1.3 Satellite Imagery***

The latest and best available satellite imagery for the study area was the Quickbird satellite imagery compiled for the Greater Hobart Area in 2005 (provided by DigitalGlobe). Images were obtained as tiles, each made up of 8192 x 8192 pixels, and spliced together as a mosaic to cover all the study sites. The imagery was of good geometric quality with a combined mean error of 0.85 m (RMS: 2.30 m; Std Dev: 2.14 m), and on analysing it with ground control points, the error was generally found to be less than a metre (Mount, pers. comm.). The images were captured at low tide.

#### ***2.2.1.4 Digital Elevation Models***

Two Digital Elevation Models (DEM) were used for the project, one of 25 x 25 m pixel size and the other 1 x 1 m. They were both obtained from Land Information Services Tasmania (LIST) and the latter, the Climate Futures LiDAR Dataset, was compiled for the Climate Futures for Tasmania project by Antarctic Climate and Ecosystems Cooperative Research Centre (ACECRC) and has a vertical and horizontal accuracy of  $\pm 25$  cm. This is the best available DEM for the study sites and is of higher quality than



the other which was derived from aerial photo interpretation and has an accuracy of  $\pm$  3.5 m.

#### ***2.2.1.5 Vector Layers***

A polygon layer for the coastline of Tasmania, provided by LIST, was used. The scale of the coastline ranged from about 1:25,000 to 1:5,000 depending on the location of the coastline. Since, the saltmarsh study sites were near highly populated areas, the scale is expected to be in the higher range.

#### ***2.2.1.6 Oblique Aerial Photographs***

Three sets of oblique aerial photographs were obtained for most of the study area. The first set was provided by Hydro Tasmania Consulting and covered limited sections in Coal River, Duckhole Rivulet and Barilla Bay areas. The second set and most comprehensive set of oblique imagery was obtained in May 2009 by the author and Ian Houshold by flying over all the study areas in the Pitt Water region. The third set, obtained from Richard Mount (taken on June 2009), covered some sections of Ralphs Bay and Pipe Clay Lagoon.

#### ***2.2.1.7 Ground-Truth Points***

A Garmin handheld Geographic Positioning System (GPS) receiver was used to record ground-truth points (GCP) on the field. The recorded GCP were then used in identifying the location and extent of a particular vegetation type in the study sites. The Garmin receivers had an error of approximately 2 to 5 m.

#### ***2.2.1.8 Field Observations and Photographs***

All study sites, except Sites 6, 11 and 17, were visited. While on the ground, records were made of aspects of the saltmarsh that were of interest to the study. These included: vegetation dominance and extent; vegetation condition; filamentous algal growth; lateral erosion; channel bank erosion; internal erosion and pond coalescing. Prints of recent imagery of the areas were taken in the field so that notes could be made of any easily visible patterns. Several of the study sites lay within private land and each landowner was contacted, initially through an introduction letter and later by phone, to obtain permission for field visits. Site 17 was not visited as the landowner was unable to be contacted. Site 6 was not visited as transport (boat or a dinghy) could not be arranged to get to the site. Site 11 was not visited as inspection of aerial photographs revealed

that the site was tidally restricted and the marsh nearly destroyed. Hence, Site 11 has been excluded from the study of vegetation change and shoreline movements.

#### ***2.2.1.9 Climatological Data***

All climatological data was obtained from the Bureau of Meteorology, Australian Government. All available records, dating back to 1958, were obtained for rainfall, temperature and wind speed. The data for wind speed and direction (wind rose) was available only from January 1995 to December 2007.

### **2.2.2 Data Processing and Analysis**

#### ***2.2.2.1 Georectification***

Both the 1965/66 (henceforth referred to as 1966) and 2008 aerial photographs were georectified in Landscape Mapper™ Version 1.4. An image to image georectification was done with the 25 x 25 m DEM and control points selected from the satellite imagery. A minimum of six control points were selected with preference given to points that were close to the area of interest, i.e. the saltmarshes. The residual mean error for 2008 aerial photographs was less than 2 m while the error for 1966 ranged from about 4 to 11 m. However, the error might not be a realistic indicator of accuracy as control points were selected keeping in mind the area of interest rather than the entire image.

The 1975 datasheets posed major problems for georectification on two levels. Firstly, the datasheets were drawn by hand, traced from unrectified black and white aerial photographs. Secondly, they were vegetation maps with no spatial features such as buildings, rocks or trees which are generally used as control points in image to image georectification process. Hence, the datasheets were not georectified with Landscape Mapper and was instead georectified “on the fly” in ESRI ArcMap™ Version 9.3. In this process, the 1966 aerial photographs were used as reference layer and control points were selected to fit each section of the marsh with the photographs and compare it with 2009 maps digitising the vegetation changes progressively for each marsh section.

#### ***2.2.2.2 Saltmarsh Mapping***

Saltmarsh mapping is undertaken for a variety of reasons, with the mapping method involved largely determined by the purpose of mapping (Kellaway *et al.*, 2009). The primary objective here was to determine the changes in the extent and type of vegetation cover of the saltmarshes in the study areas from 1975 to 2009. This required the

adoption of the mapping method employed in 1975 (Glasby, 1975; Kirkpatrick, pers. comm.). The vast improvement in mapping technologies and the different resources that were available constrained the extent to which the 1975 mapping processes were adapted. However, most importantly, the mapping rules remained the same.

### **Mapping Rules**

Primarily, saltmarshes were defined to be “tracts of land tidally connected to the sea and covered with phanerogamic halophytic vegetation comprised of herbs, shrubs, grasses, sedges and rushes, and including the associated tidal channels, salt flats and marsh pools.” This definition excludes freshwater aquatic sedgeland and rushland (ASF), as defined by Harris and Kitchener (2005), which can occur towards the terrestrial side of saltmarshes. The main rules/guidelines for mapping the extent of the saltmarshes included setting the landward and seaward limits of the marsh. The seaward limit of the saltmarsh (shoreline) was determined by saltmarsh vegetation (usually *Sarcocornia quinqueflora* or *Juncus kraussii*) boundary. The landward limit was usually determined by the halophyte-glycophyte boundary, usually identified by the inland limit of *Austrostipa stipoides* or *J. kraussii*. In the upper estuary, the limit was set by the presence of *Phragmites australis*, a common reed found in the fluvial systems of Tasmania. Vegetation dominated by this taxon was not considered to be saltmarsh.

Vegetation was mapped by the dominant species in the tallest stratum. An identified dominance group was required to have greater than 10% of foliage protective cover over an area large enough to be represented on the map at a scale of 1:10,000 as an individual patch or polygon. Based on this guideline, 30 dominance groups (or vegetation communities) were identified and used in the mapping. In addition to the 30 dominance groups, two more groups were used to represent bare ground devoid of any vegetation and non-saltmarsh vegetation that occurred within or closely associated with the dominance groups. These 32 groups have been considered to be major structural components of the saltmarshes.

### **Mapping Process**

Mapping largely followed the three step process for the “extent of cover analysis by pedestrian survey” method detailed by Kellaway *et al.* (2009). However, some changes were necessitated due to reasons pertaining to resource availability and project aims.

The process employed here included five main data sources: aerial photographs (both recent and historical); satellite imagery; oblique aerial photographs; and the ground truth data. These inputs were all used as “layers of evidence” in the mapping process.

The first step in the process was preliminary GIS mapping where aerial photographs and satellite imagery were used to map the extent of the saltmarsh and delineate structural components in ArcMap. The use of both aerial photographs and satellite imagery assisted in identifying colour differences caused by canopy shading in the saltmarsh-upland boundary by tall terrestrial plants such as *Eucalyptus* spp. and *Allocasuarina verticillata*. The satellite imagery was extremely useful in identifying the boundary between dominance groups of varying height. The aerial photograph was ineffective in identifying the boundary between *T. arbuscula* and *S. quinqueflora* dominance groups while satellite imagery proved to be very effective. The main advantage of the aerial photographs was in identifying bare patches and differentiating dominance groups which had strong colour variations (such as vegetation dominated by *Disphyma crassifolium*).

Aerial photographs and satellite imagery, while helpful in identifying the boundaries of major patches, were not at a suitable scale for identifying and delineating all structural components consistently at a scale of 1:10,000. Hence, the data extracted from preliminary GIS mapping process was supplemented with extensive ground-truthing by both pedestrian and air surveys. Pedestrian surveys were carried out with preliminary maps prepared from GIS mapping. The entire length and breadth of each site was surveyed to verify and correct boundaries, and identify patches that were missed in the GIS mapping. A handheld GPS unit was used to record GCP at each patch that was not clearly identifiable on the aerial and satellite images. The number of GCP collected for a patch depended on its size, and for big patches, points were collected along the boundary at regular intervals. However, given the enormous area of the saltmarshes and limited personnel and time, the possibility of overlooking patches or misrepresenting their boundaries was high. Hence, oblique photographs were acquired for most sections of the saltmarshes. Oblique photographs were obtained both on the ground, on vantage points wherever they were available, and from the air. Oblique photographs, especially those taken from the air, provided excellent views of the patches and their boundaries

and have considerably improved the mapping accuracy. The final vegetation map was prepared after up to four iterations considering several layers of evidence.

The mapping process concurs well with the recommendations made by Wilton (2002) for mapping saltmarshes. Mapping for habitat change was done at a scale of 1:10,000 or larger. Distortion errors were corrected in aerial photographs with georectification which used a minimum of six GCP. The community boundaries were digitised on-screen, thus eliminating errors associated with hand-drawn boundaries. The aerial photos were scanned at a resolution of more than 300 dpi.

#### ***2.2.2.3 Changes in Extent***

Changes in the saltmarsh extent were calculated both on the seaward and landward side. Two input layers were prepared. The first one was the digitised boundary of the 1966 extent of the saltmarshes and the second, the 2005/08 boundary. Aerial photographs have several limitations in accurately identifying saltmarshes with respect to identifying the saltmarsh extent (Kellaway *et al.*, 2007). Hence, the 1975 saltmarsh maps were used to correct the boundaries digitised on the 1966 aerial photographs. The recent boundary was digitised from the 2005 satellite imagery and corrected with 2008 aerial photographs (hence mean yearly shoreline changes have been calculated between 1966 and 2006). Satellite imagery was used to digitise the boundaries as they are expected to have less distortion error than aerial photographs which have the cumulative error of the satellite imagery and the georectification process. Correction with 2008 aerial photography was done in places where the tall high marsh plants created a shadow effect on the seaward boundary in the satellite image. The accuracy of changes is much higher on the seaward side as compared to the landward side. This was because the differentiation of saltmarsh vegetation from terrestrial vegetation in the 1966 aerial photographs was difficult in some places. Erosion and accretion were calculated at 607 sampling points digitised on the 1975 coastline at 50 m intervals (see Appendix I).

#### ***2.2.2.4 Changes in Vegetation***

A vegetation change polygon layer was digitised by comparing the 1975 and 2009 vegetation maps. In this process, each 1975 polygon was compared with 2009 polygon, and where changes of vegetation had occurred, a “change polygon” was digitised covering the area of change. In places where the change in area and configuration of the 1975 and 2009 polygons was limited while the dominance group remained the same, the

change was omitted as mapping inconsistencies and observer bias might have been involved in misrepresenting the patch. In some cases, 1975 polygons on the seaward side of the saltmarshes had eroded (becoming intertidal mudflats) and were not recorded as vegetation change. Kellaway *et al.* (2009) identified that one of the issues of time-series mapping is that, due to the improvements in mapping technology, number of patches identified will be more with each new mapping effort. While this was true in some sections of the marsh, for others, the number of patches recorded was the same as in 1975 and in some instances, lesser. Mapping inconsistencies and observer bias are possible contributors to these changes. Hence to avoid the incorporation of these inconsistencies into change analysis, all smaller patches were not considered to be changes. In most of the saltmarshes, grasses/sedges/rushes form a border on the terrestrial side of the marsh. In some cases, these borders were wide enough to be mapped and in others, they were only a thin line of less than a metre in breadth. The 1975 maps show distinct borders of grasses/sedges/rushes in almost all the marshes. It is considered that the actual sizes of these patches were over-represented in the map in an effort to indicate that they existed on the ground.

#### ***2.2.2.5 Wave Exposure Analysis***

The wave exposure analysis was done using the GIS based cartographic wave exposure model developed by Pepper (2009). The model was selected as it was a fast and easy to use application that can be built with ArcMap and customised based on project needs. The data inputs to the model involved a point file, polygon file and wind speed/directional information. The point file was a collection of 607 sampling points digitised on the 1975 coastline at 50 m intervals. The wave exposure model calculated the fetch length between each sampling point and the nearest potential wave blocking obstacle for every 7.5 degrees around the point (48 fetch lines; Figure 2.22). The polygon file, which was the digitised coastline of Tasmania, was used in the model as the input for wave blocking obstacle. The calculated fetch length was then weighted by the wind speed and direction information (for 8 compass directions) to calculate the relative wave exposure for each sampling point along the saltmarsh shoreline. Since the relative wave exposure is a dimensionless value, it is henceforth called the Wind-Fetch Index.



The Wind-Fetch Index was compared with the advancement and retreat data to examine the degree of correlation between the two. Data for advancement and retreat was calculated at each of the 607 sampling points which were used in the wave exposure model and organised into four groups, one for advancement and three for retreat. A statistical test was run using one-way analysis of variance (ANOVA) in Minitab<sup>®</sup> Version 15 to test whether four groups – advancement; 0-5 m retreat; 5-10 m retreat; and >10 m retreat – differed in their mean Wind-Fetch Index.



Figure 2.22. An illustration of fetch length calculation with Pipe Clay Lagoon as an example. The image on the left shows saltmarsh shoreline vulnerability to wave exposure (measured in terms of Wind-Fetch Index), with bigger circles indicating higher exposure. The image on the right shows the fetch lines for each sampling point on the shoreline, with three points highlighted for illustration purpose. Software source: Pepper (2009).

#### 2.2.2.6 Inundation Modelling

The inundation modelling was done in ArcMap using the LiDAR DEM and with a high-end sea level rise estimate of 1.1 m by the year 2100 (selected based on inputs from Mount, pers. comm. and Sharples, pers. comm.). The modelling was carried out by Michael Lacey using parameters defined by the author for this study. The model output represents the probability of the areas inundated by the flood tide at mean high water mark with the estimated sea level rise. The probabilities were derived from the known

errors in the input datasets, which includes data capture and processing errors in the LiDAR DEM (Mount, pers. comm.). Maps were prepared to represent the inundated areas for five different probabilities (see Appendix II).



## Chapter 3: Changes in the Extent of Saltmarshes

### 3.1 Introduction

Saltmarsh development and extent (morphodynamics) is governed by four main “forcing factors” as described by Allen (2000; Chapter 1). They include: the influence of the sea and wave energy; the amount and quality of sediment available; plant productivity; and autocompaction. In addition to these forcing factors, human influences and tectonic subsidence can be said to form the major “agents of change” in determining the morphology and extent of a given saltmarsh (Figure 3.1). Changes in extent result when the change agents exert pressure on the saltmarsh ecosystem. Changes can occur both on the seaward and landward side. On the seaward side, the two main changes are advancement or retreat of the seaward boundary of the saltmarsh, usually due to the process of sediment accretion or erosion respectively. On the landward side, increased landward penetration of tidal water may facilitate the transgression (or migration) of the saltmarsh plant communities inland, or increased sedimentation (by aeolian or fluvial processes) can exclude parts of the marsh from tidal inundation and make way for non-saltmarsh plants to take over. Both the seaward and landward areal changes can be substantially affected by human activities such as raising tidal barriers and dumping fill on the marsh.

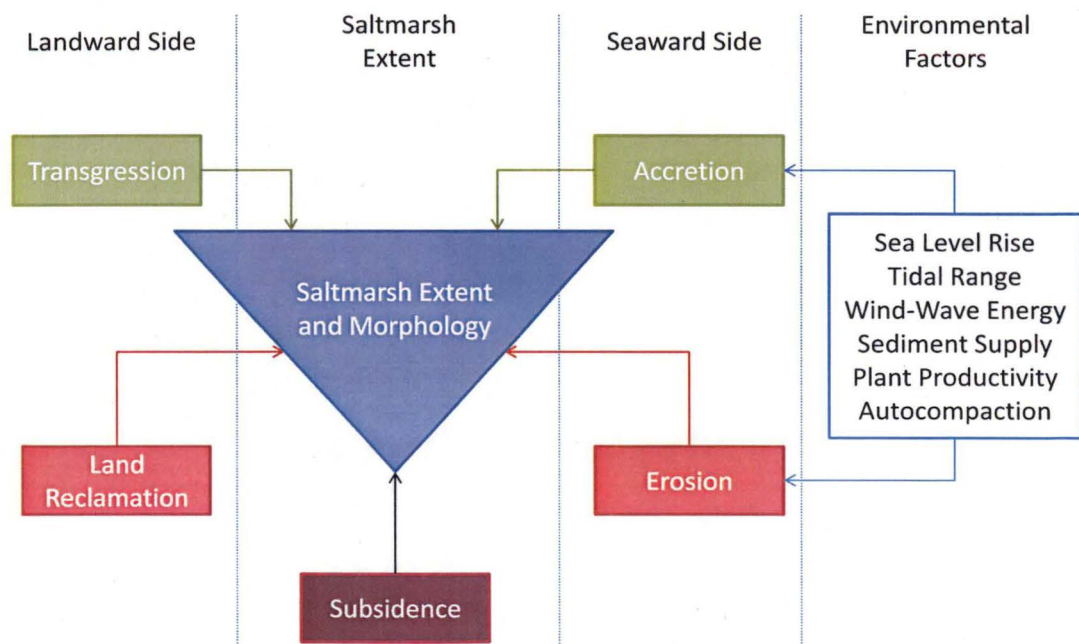


Figure 3.1. Major processes and environmental factors governing the extent and distribution of saltmarshes. Environmental factors (forcing factors) are as described by Allen (2000).

The aims of this chapter are to discuss the agents responsible for change in the extent of saltmarshes, document the changes observed in the saltmarshes of the study area, determine the correlates of change and predict future changes.

### 3.1.1 Agents of Change

#### 3.1.1.1 Human Influences

Human influences on the saltmarsh are numerous and are both direct and indirect (Adam, 2002). Direct human impacts range from the defoliation and soil compression associated with a simple bicycle ride or dog-walking to the major impacts of soil dumping for land extension. Indirect effects occur when human actions inadvertently (or unintentionally) have a negative impact on the ecology of the marsh. For example, damming a river that has saltmarshes downstream might reduce the sediment inputs required for the marsh to compensate sediment losses caused by erosion (Zedler and Adam, 2002).

The types of human influences on saltmarshes vary across different parts of the world. A summary of the human effects on saltmarshes has been provided by Kennish (2001) for the American context, Doody (2008) for the European context; and, Laegdsgaard *et al.* (2009) for the Australian context. The major human influences on saltmarshes in Tasmania were detailed by Kirkpatrick and Glasby (1981). They include:

1. **Landfill:** Several saltmarshes have been used as rubbish dumps, and regarded as wastelands and converted for “higher” purposes such as roads and farmland.
2. **Catchment modification:** Dams have been built on numerous rivers and creeks that have saltmarshes in their estuaries and creek mouths. These dams reduce the amount of sediment inputs available for the marsh and also change the frequency and flow of freshwater.
3. **Fire:** While fire is uncommon in saltmarshes dominated by succulents, when they do occur, they can eliminate the succulent shrubs from the marsh (as in Orielton Lagoon).
4. **Grazing:** Sheep and cattle graze and trample on the marsh reducing the community composition, abundance and growth of the saltmarsh plants.
5. **Off-road vehicles:** recreational use of off-road vehicles on the saltmarsh causes major losses in plant cover.

6. **Exotic species:** some exotic species tend to replace the native saltmarsh plants (such as *Juncus acutus* and *Spartina anglica*) while others occur on the terrestrial fringe (such as *Plantago coronopus*). Certain other species occur where disturbance creates bare spaces occur within saltmarshes for weeds to colonise (such as *Senecio elegans*).

In addition to these, tidal restriction (or drainage) and pollution can also be considered to be important human influences on Tasmanian saltmarshes. Gouldthorpe (2000) in his study of the effects of drainage and grazing on some south east Tasmanian saltmarshes underlined the detrimental effects of drainage and grazing by referring to them as the two most serious impacts for saltmarshes. Pollution caused by agricultural fertilizers can cause widespread deterioration of the saltmarsh environment available for both the flora and fauna (Laegdsgaard *et al.*, 2009). Human disturbances on saltmarshes can cause loss or simplification, resulting in losses in their functions and services (Laegdsgaard, 2006).

#### **3.1.1.2 Subsidence**

The increase in the volume of the oceans from melting of glaciers and warming has caused many parts of the coasts, especially in temperate latitudes, to subside in isostatic movements (Woodroffe and Davies, 2009). For example, the coast of south-east England has been subsiding primarily due to isostatic adjustment causing erosion and loss of saltmarshes along the coast (Boorman, 1999). Saltmarsh loss due to subsidence have also been recorded on the Atlantic coast of the United States of America (Phillips, 1986) and Italy (Day *et al.*, 1998), among other places.

#### **3.1.1.3 Climate Change and Sea Level Rise**

The coastal saltmarsh exists in a close relationship with the sea and is regarded more as “an intertidal profile, rather than as a self-contained landform” (Pethick, 1992, p. 41). Hence, the coastal processes play a major role in shaping the saltmarsh and any changes in these processes will have pronounced effects on the morphology of the marsh. Recently, the relative rise in the mean sea level has had a profound effect on pattern and process in the coastal zone (Pethick, 1993; Day *et al.*, 2000; FitzGerald *et al.*, 2008). Numerous studies have shown that sea level rise (both eustatic and isostatic) has been causing severe erosion of coastal saltmarshes (e.g. Phillips, 1986; Pye, 1994; Wrayf *et al.*, 1995; Day *et al.*, 1998; Cooper *et al.*, 2001; Schwimmer, 2001; Hartig *et al.*, 2002;

Morris *et al.*, 2002; van der Wal and Pye, 2004; Wolters *et al.*, 2005). Further rise in sea level could exacerbate losses of coastal saltmarshes through continued erosion and drowning, where sedimentation is not able to keep with erosion (Orson *et al.*, 1985; Schwimmer and Pizzuto, 2000).

Of the major factors that drive erosion in saltmarshes, wind generated wave energy is considered to be the most important (D'Alpaos *et al.*, 2009). Harris *et al.* (2002, p. 868) have suggested that “[g]eomorphic changes resulting from an increase in wave power are likely to be most obvious for tide-dominated environments” in the Australian context. Wave energy/power is strongly correlated with wave exposure, which is primarily a measure of aggregated fetch length (the sum of the distance of open water in a defined number of evenly spaced directions), wind strength and directional variability, with a frequency distribution of water stages included for tide dominated environments (Allen and Pye, 1992). Wave exposure has been shown to be significantly correlated with the erosion of saltmarsh shorelines (Day *et al.*, 1998; Schwimmer, 2001) and an increase in relative SLR is expected to increase the intensity of the wave exposure and hence the erosional stress on the marsh shoreline (Pethick, 1992, 1993). Furthermore, global warming is expected to increase the wave heights and cause increased storminess, thus posing a major threat to coastal ecosystems (Gulev and Hasse, 1999; Grevemeyer *et al.*, 2002).

Several models have been developed to predict the response of the coastal saltmarshes to a rise in sea level (e.g. Orson *et al.*, 1985; Schwimmer and Pizzuto, 2000; Simas *et al.*, 2001; Kirwan and Temmerman, 2009). The models of Orson *et al.* (1985) and Schwimmer and Pizzuto (2000) suggested that the coastal saltmarsh will retreat by erosion or eventually drown if the sedimentation rate is not able to keep up with the sea level rise. Simas *et al.* (2001) associated the elevation of the marsh with its vulnerability to sea level rise, and suggested that areas with high tidal ranges are less susceptible to changes in sea levels. Kirwan and Temmerman (2009, p. 6), in their analysis of the response of the coastal marsh to sea level rise, indicated that it “takes on the order of 100 years for a marsh to adjust to an increase in the rate of sea-level rise, and that during a continuous sea-level acceleration, accretion rates lag behind sea-level rise rates by about 20–30 years.”

In general, with a rise in sea-level, the marsh is expected to accrete vertically as it is eroded laterally. However, when the rate of vertical accretion (by both allochthonous and autochthonous means) is not able to keep with the rising sea level, the marsh is expected to become open water or a lower intertidal area (FitzGerald *et al.*, 2008). In places where there are no physical barriers, whether artificial or natural, that prevent the marsh from migrating inland, a rise in sea level is expected to see the transgression of the marsh inland (Donnelly and Bertness, 2001). In some cases, such inland migrations have been facilitated by the removal of the barriers, and have resulted in pioneer saltmarsh community being formed (Boorman, 1999).

## 3.2 Results

### 3.2.1 Shoreline Advancement/Retreat

A major part of the saltmarsh shoreline has retreated landward over the study period. Of the 607 points sampled on the seaward boundary of the saltmarsh on the 1966 aerial photographs, 459 points indicated retreat by 2006 while 148 indicated seaward advancement. Within the sampled points that indicated retreat, the degree of retreat varied greatly, from less than a metre to more than 90 m. The average retreat in 41 years was 10.4 m, a mean of  $0.26 \text{ m yr}^{-1}$  (Table 3.1). Orielton Lagoon had the highest retreat of 37.2 m at almost a metre lost every year and three sampled points with a retreat of above 90 m. This retreat was probably caused by changes to the tidal regime in the lagoon consequent upon the opening of the lagoon to the Pitt Water estuary, and is therefore exceptional. Excluding Orielton Lagoon, the mean retreat of other sites ranged from 2.4-8.0 m (overall mean of 5.1 m) at means of  $0.06\text{-}0.20 \text{ m yr}^{-1}$  (overall mean rate of  $0.13 \text{ m yr}^{-1}$ ). Ralphs Bay, Barilla Bay and Iron Creek had above average rates of retreat with Coal River/Duckhole Rivulet and Pipe Clay Lagoon having comparatively lesser rates of retreat.

**Table 3.1. Mean retreat rate of the shoreline at the retreating sampled points by study area.**

Area Name	Mean Retreat (m)	Mean Retreat/ Year (m)	Standard Deviation (m)	Number of Points Sampled
Coal River/ Duckhole Rivulet	2.4	0.06	3.5	268
Barilla Bay	6.0	0.15	5.9	109
Orielton Lagoon	37.2	0.93	32.4	20
Iron Creek	5.7	0.14	4.4	53
Pipe Clay Lagoon	3.2	0.08	2.8	84
Ralphs Bay	8.0	0.20	9.7	73
<b>Total</b>	<b>10.4</b>	<b>0.26</b>	<b>9.8</b>	<b>607</b>
<b>Total excluding Orielton Lagoon</b>	<b>5.1</b>	<b>0.13</b>	<b>5.3</b>	<b>587</b>

A considerable area of saltmarsh was lost to the sea over the study period. The area lost was estimated to be 173,651 m<sup>2</sup>, which was approximately four times as much as saltmarsh area gained (43,653 m<sup>2</sup>; Table 3.2). Excluding Orielton Lagoon from the analyses provided a higher ratio of gain to loss with one square metre gained for every 4.7 m<sup>2</sup> lost. The net loss was estimated to be 129,998 m<sup>2</sup>, at a rate of about 3,250 m<sup>2</sup> a year after discounting the surface area gained through advancement. All study areas lost more than 10,000 m<sup>2</sup>. The largest changes were in Orielton Lagoon, with about 21,930 m<sup>2</sup> of area gained and twice as much lost. However, Barilla Bay stands out as the area with the highest retreat, 36,700 m<sup>2</sup>, compared with the amount of advancement, 1,159 m<sup>2</sup>. In other words, for each square metre of advancement, there was nearly 32 m<sup>2</sup> of retreat.

**Table 3.2. Changes in saltmarsh extent along the seaward edge of the saltmarsh.**

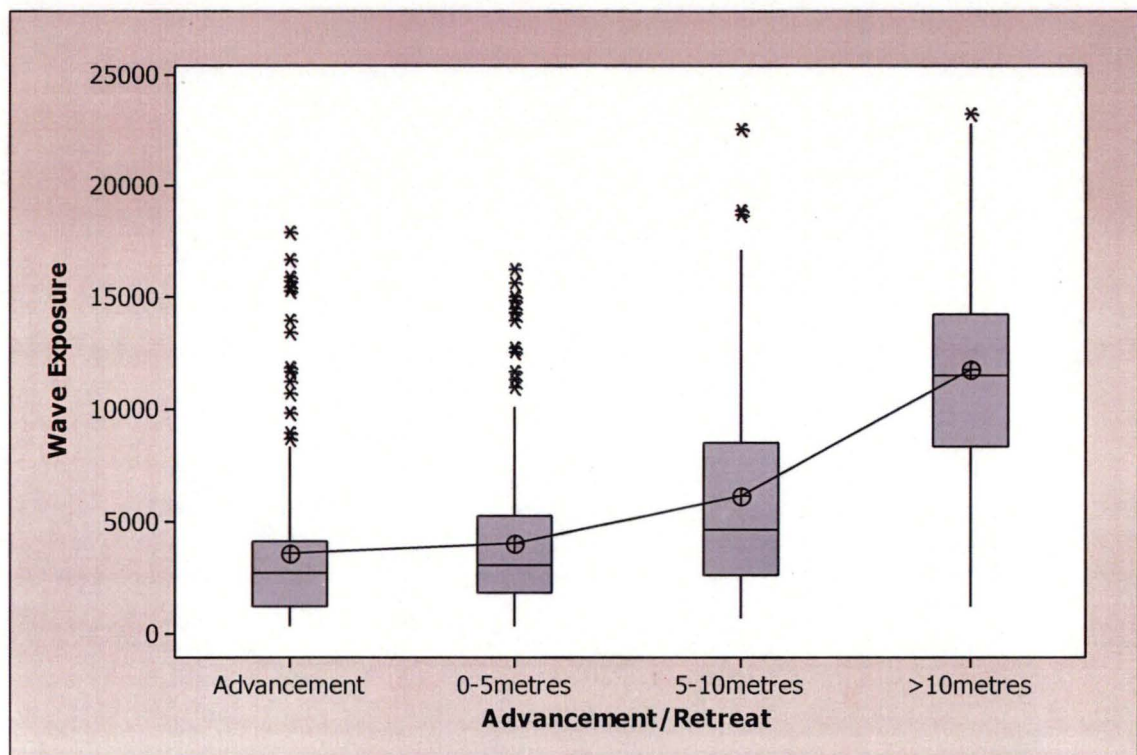
Area Name	Advancement (sq. m)	Retreat (sq. m)	Net Loss (sq. m)	Percentage of the Total Loss	Ratio of Gain to Loss
Coal River	8,784	25,397	16,613	12.8	1:2.9
Duckhole Rivulet	2,874	14,912	12,038	9.3	1:5.2
Barilla Bay	1,159	36,700	35,541	27.3	1:31.7
Orielton Lagoon	21,930	41,953	20,023	15.4	1:1.9
Iron Creek	3,000	16,933	13,933	10.7	1:5.6
Pipe Clay Lagoon	2,122	13,154	11,032	8.5	1:6.1
Ralphs Bay	3,784	24,602	20,818	16.0	1:6.5
<b>Total</b>	<b>43,653</b>	<b>173,651</b>	<b>129,998</b>	<b>100</b>	<b>1:4</b>

### 3.2.2 Wave Exposure Analysis

There is a positive correlation between the degree of wave exposure and advancement/retreat (Figure 3.2). The four advancement/retreat classes are strongly related to wave exposure ( $F = 72.95$ ; d.f. = 3;  $P < 0.001$ ;  $r^2 = 27.29\%$ ). The advancement class is not significantly different from the 0-5 m retreat class, but differs significantly from all others. All retreat classes vary significantly from each other. Furthermore, a clear threshold is noticeable between the >10 m retreat class and the other classes indicating that as the wave exposure increases above the 7000-8000 mark (a



dimensionless index, referred to as Wave-Fetch Index), there is an increase in the degree of retreat.



**Figure 3.2:** Box plot showing the relationship between relative wave exposure (expressed as a dimensionless Wave-Fetch Index) and shoreline movement (1966-2008). Boxes represent the data range between the first and the third quartile (25-75% of the data). The middle horizontal line represents the median value, and the circle with a cross the mean. Whiskers extend to the maximum and minimum data points within 1.5 box heights. Observations that lie beyond the range of the whiskers (outliers) are represented by stars.

Parts of the saltmarsh shoreline (outliers in Figure 3.2) did not fit the wave exposure-retreat relationship. These outliers occurred in six main sections of saltmarsh shoreline (Table 3.3). On further inspection of the outliers, it became apparent that at any point on the saltmarsh shoreline other forces (such as tidal currents and longshore drift) might be persistent enough to counteract the wave energy and facilitate accretion (Sharples, pers. comm.). In the case of the spit in Iron Creek (Site 16a, Figure 3.3), longshore drift could be involved in the process of sediment deposition and elongation of the spit (Houshold, pers. comm.). Hence, despite very high wave exposure, the end of the spit had advanced and elongated (Figure 3.3). The outliers in Barilla Bay and Pipe Clay Lagoon were possibly due to the effect of bathymetry as both the outliers were near shallow intertidal areas. Three sets of outliers were identified in Ralphs Bay Site 21. One was on the artificial spit, which had considerable wave erosion on the edge of the spit, yet had accretion near the base, which was acting as a sediment trap. Two other sections in Site



21 were outliers because of dumping from the adjacent road works and the formation of a secondary marsh.

Table 3.3. Major sets of erosion outliers and possible reasons.

Outlier No.	Area Name	Site No.	Possible Reasons
1.	Iron Creek	16a	Spit formation
2.	Barilla Bay	14	Bathymetry
3.	Ralphs Bay	21	Sediment trap
4.	Ralphs Bay	21	Dumping from road works
5.	Ralphs Bay	22	Secondary marsh formed
6.	Pipe Clay Lagoon	20	Bathymetry

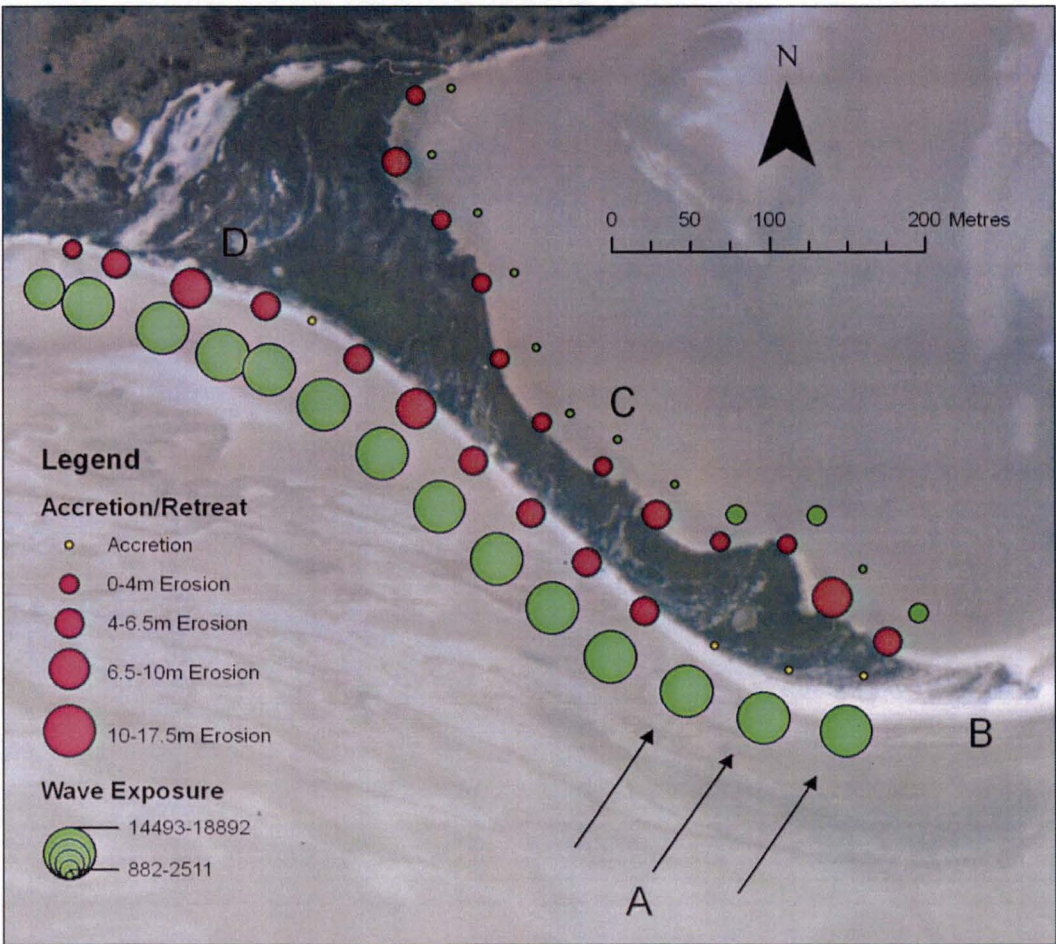


Figure 3.3. Accretion/Retreat-Wave Exposure relationship for the Iron Creek spit (Site 16a). A – indicates the outliers where there has been accretion despite a very high wave exposure; B – Lengthening of the spit (also see Plate 3.1); C – Erosion on both sides of the spit has effectively reduced the width of the spit; D – a high wave exposure has deposited a layer of sand/shell sheet over the marsh (see Figure 3.14).

### ***3.2.2.1 Change in Wind Climate***

An analysis of the wind speed data recorded over the 50 year time period (1959 to 2008) covering the time period of this study shows a stochastic rise and fall of annual wind speed over the years (Figure 3.4). However, the incidence of low intensity winds (of <10 km/h) decreased considerably from around 1990 onwards, with the mean average wind speed rising to over 15 km/h. The rise in wind intensity has been quite marked since 2002 with more than 75% of the observations recording a wind speed of >10 km/h. The change in wind speed over the years is striking when the data for the 10 years before the current study (1999-2008) is compared with a period of 10 years before the previous study (1966-1975) with a significant shift in wind strength between the two decades (Figure 3.5).

### ***3.2.3 Patterns of Shoreline Advancement/Retreat***

Some general patterns of changes have been identified by aerial photographic interpretation and direct field observations. They are:

1. moderate to severe retreat/erosion along the saltmarsh shorelines that are highly exposed to wind-generated waves;
2. extension and lateral expansion of tidal channels;
3. migration and transgression of shell ridge and sand sheets over the saltmarsh;
4. areas of advancement/accretion and changes in the coastal geomorphology.

#### ***3.2.3.1 Patterns of Edge Erosion***

Some areas of shoreline retreat were clearly visible with direct comparison of the aerial photographs. In most cases, shorelines that had experienced a greater retreat also had a high degree of exposure to wind-generated waves. An example of this relationship is illustrated in Figure 3.6 where there was severe erosion on northern and eastern shorelines of the eastern end of Site 12, which was highly exposed to wind-generated waves, as compared to the relatively lesser erosion on the southern shoreline, which was less exposed to wind-generated waves. A closer examination of the eastern shoreline of Site 12 shows that the 1966 shoreline had retreated over 25 m and a tidal channel, then several metres inland, became part of the shoreline by 2005 (Figure 3.7).



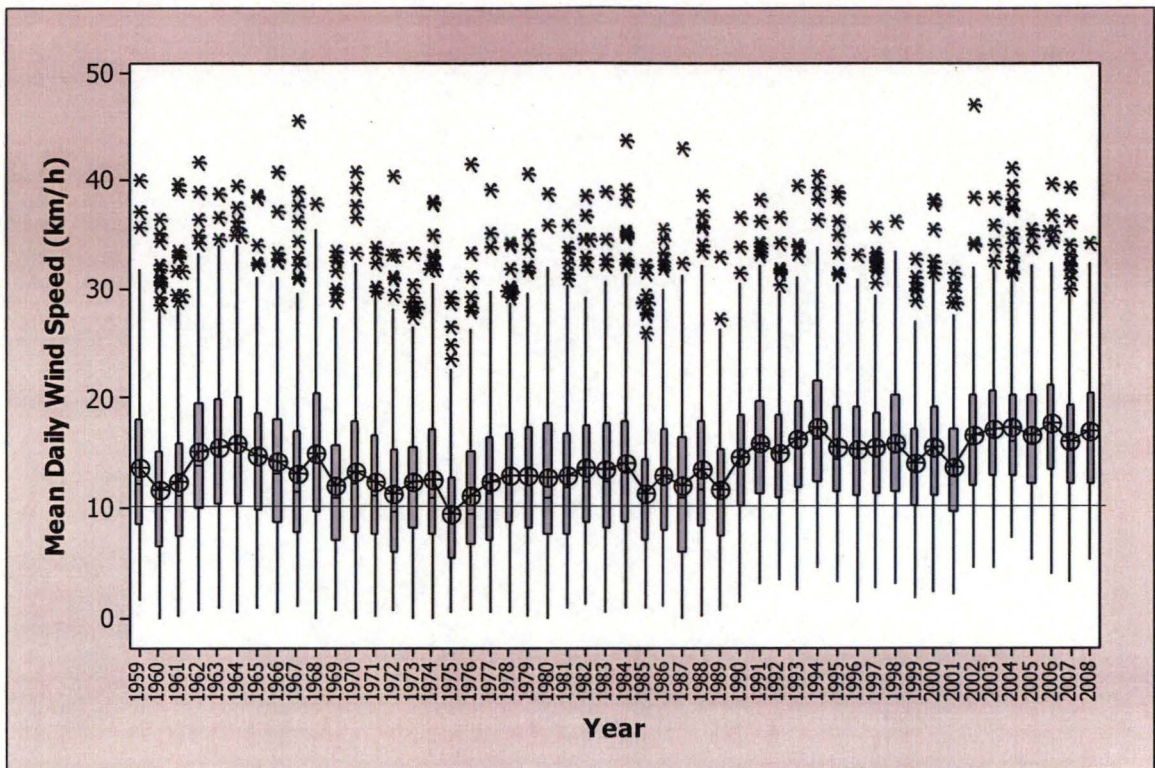


Figure 3.4. Box plot of mean daily wind speed (in km/h) recorded at Hobart Airport. Hobart Airport is within 15 kilometres of all the study sites. Refer to Figure 3.2 for the key to the symbols used.

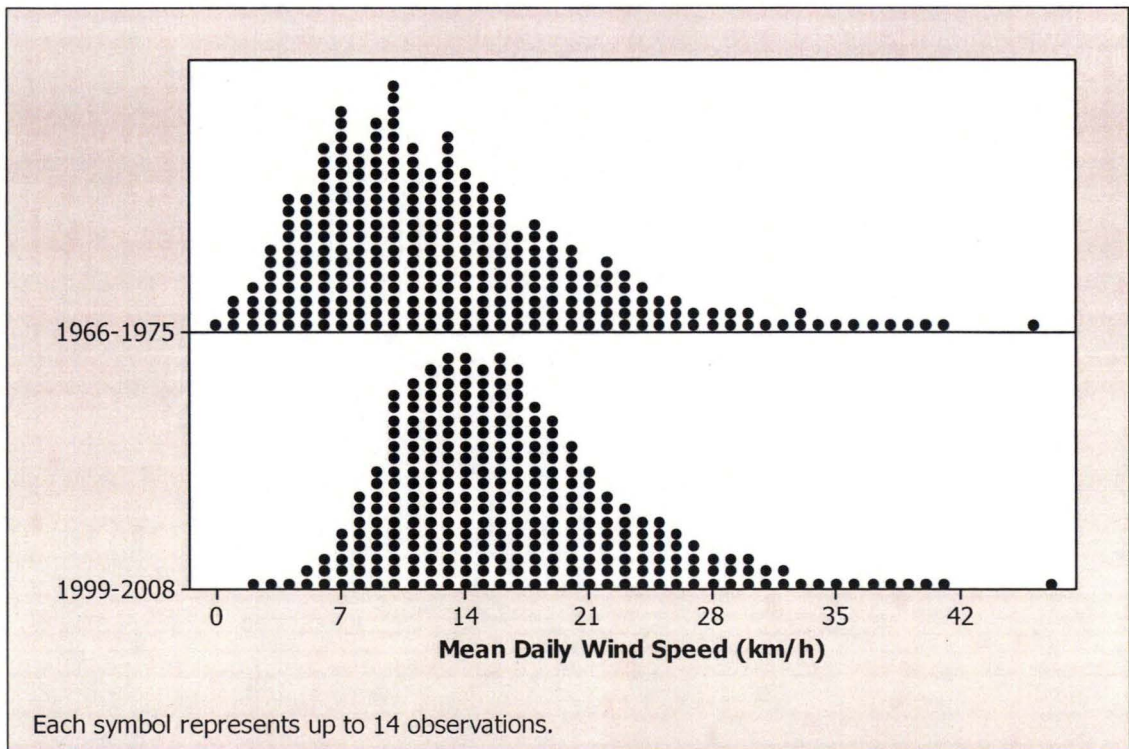


Figure 3.5. A dot plot showing the mean daily wind speed (in km/h) recorded over two decades, 1999-2008 and 1966-1975, preceding the current and previous study.



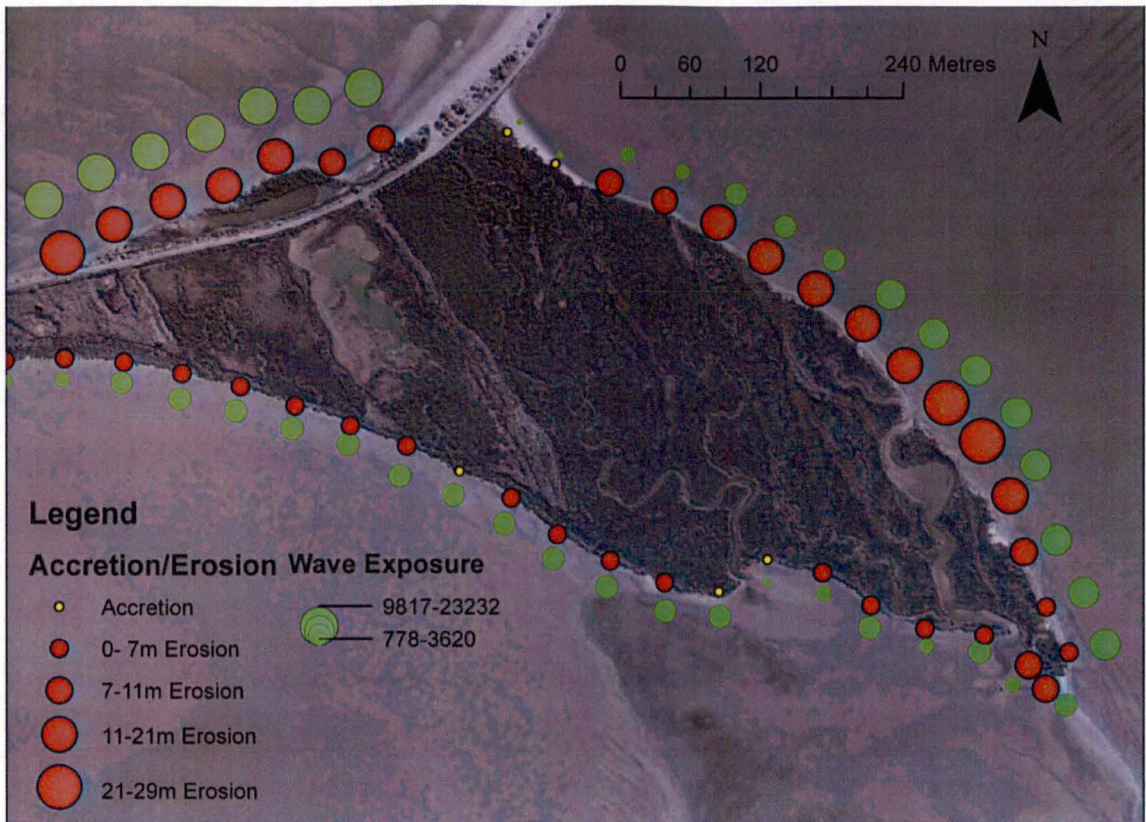


Figure 3.6. Accretion/erosion and wave exposure in the eastern end of Site 12 in the Barilla Bay area. The size of the circles represents the intensity of erosion and wave exposure and it can be noted that generally areas of high wave exposure have suffered high rates of erosion.

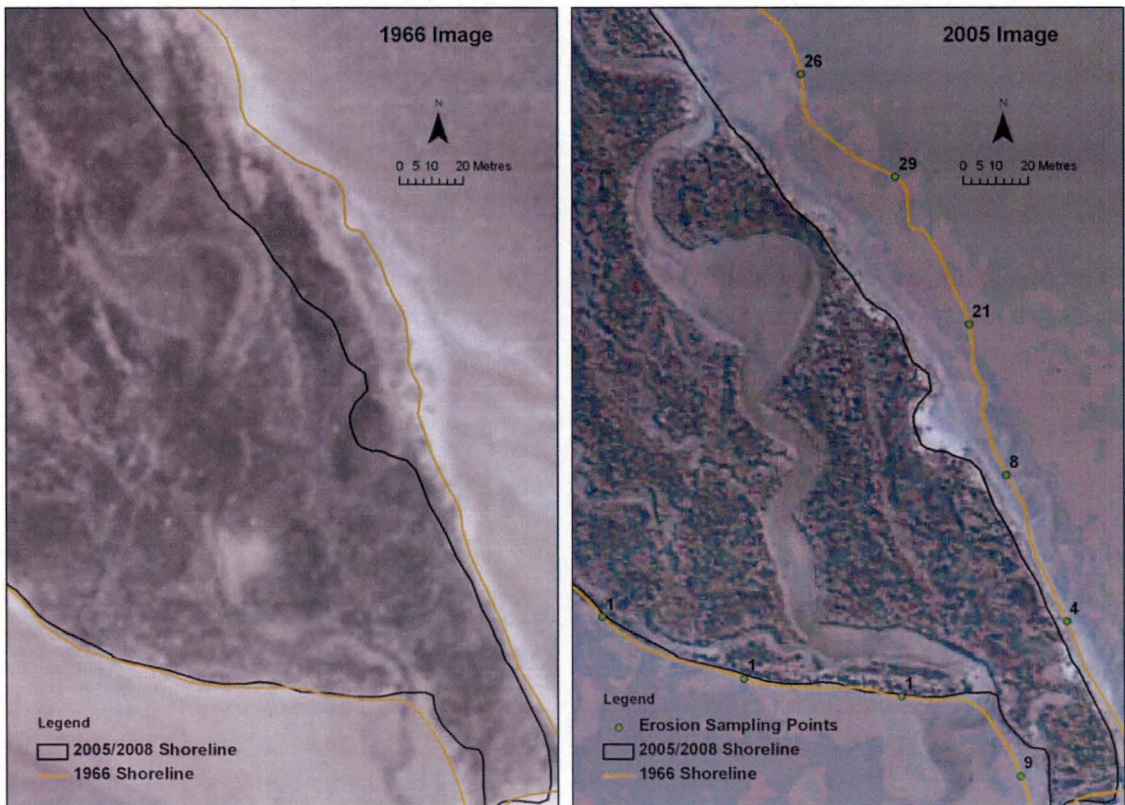


Figure 3.7. A closer view of the south-eastern end of the marsh indicates severe erosion (up to 30 m) with shoreline encroachment into the side of a tidal channel effectively separating a section the marsh.

Direct field observations of the shoreline retreat revealed a general pattern of retreat caused by erosion. The rate and the process of erosion appeared to be largely governed by substrate type, vegetation and the height of the marsh surface above mean sea level. Where the marsh surface is high relative to mean sea level, erosion caused the formation of microcliffs (or erosion scarps) which were up to a metre in height (Figure 3.8). In some places, the eroded shoreline showed a pattern of necks and clefts, usually with the root system of succulent shrubs (typically *T. arbuscula*) or grasses (typically *A. stipoides*) holding the neck together by providing some resistance to erosion (Figure 3.9). It was noted that grasses, with their caespitose root systems, were more resistant to erosion than the succulent shrubs (which have a taproot). In some cases, the necks were noticed to have toppled (or slumped) due to undercutting of their substrate. Undercutting was mostly observed under the rootmat where the substrate is not held together by the root system.

In marshes situated at a low elevation relative to the mean sea level, erosion primarily appeared to be by wave scouring at the seaward edge of the marsh. In such cases, the erosion scarp was less well defined. Field observations indicated an erosion process where waves begin to erode the marsh surface exposing and then removing the roots from the substrate. As the roots are removed, the substrate provides less resistance to the wave energy and eventually gets washed away thereby killing the uprooted plants and resulting in a landward movement of the marsh shoreline (Figure 3.10). In places where the marsh was only a thin fringe separating the sea and a natural or artificial landward barrier (such as roads, levees or sharply rising upland), persistent erosion has reduced the marsh extent in a process known as “coastal squeeze” (e.g. Cooper *et al.*, 2001). This was observed in Ralphs Bay and Pipe Clay Lagoon where roads immediately behind the marsh acted as barriers to landward migration (Figure 3.11). In some parts of Duckhole Rivulet and Coal River sharply rising uplands were responsible for coastal squeeze.





Figure 3.8. Scarps on the saltmarsh edge indicating severe wave erosion on the eastern shoreline of Site 12 in Barilla Bay.



Figure 3.9. Pattern of necks and clefts formed due to erosion on the saltmarsh shoreline. Note the roots of the grass (*Austrostipa stipoides*) holding the substrate together from erosion. However, the necks can be seen to be undercut, a gradual process which will eventually cause slumping and erosion.





Figure 3.10. Erosion on low elevation marshes where the waves scour the shoreline eroding the substrate, killing the plants and causing shoreline retreat.



Figure 3.11. This area in Ralphs Bay was once a fringing marsh which was 'squeezed' by the waves and the road barrier behind. All that remains now is a lone survivor (*Suaeda australis*) clinging on to a pipe which was closer to the original elevation of the marsh.



### 3.2.3.2 Expansion of Tidal Channels

A frequently observed change in the saltmarsh physiography at the study sites was the lengthening and widening of the tidal channels. Figure 3.12 and Figure 3.13 are of Site 13 in Barilla Bay which, besides showing shoreline retreat over the study period, also shows the increase in the landward penetration and width of the tidal channels. On ground, the tidal channels showed clear signs of erosion due to slumping of the channel banks (Figure 3.13). Since well developed tidal channels were found only on sand/silt substrate in relatively large marsh platforms, this phenomenon was widely recorded in Coal River, Duckhole Rivulet and Barilla Bay as compared to the other study areas.

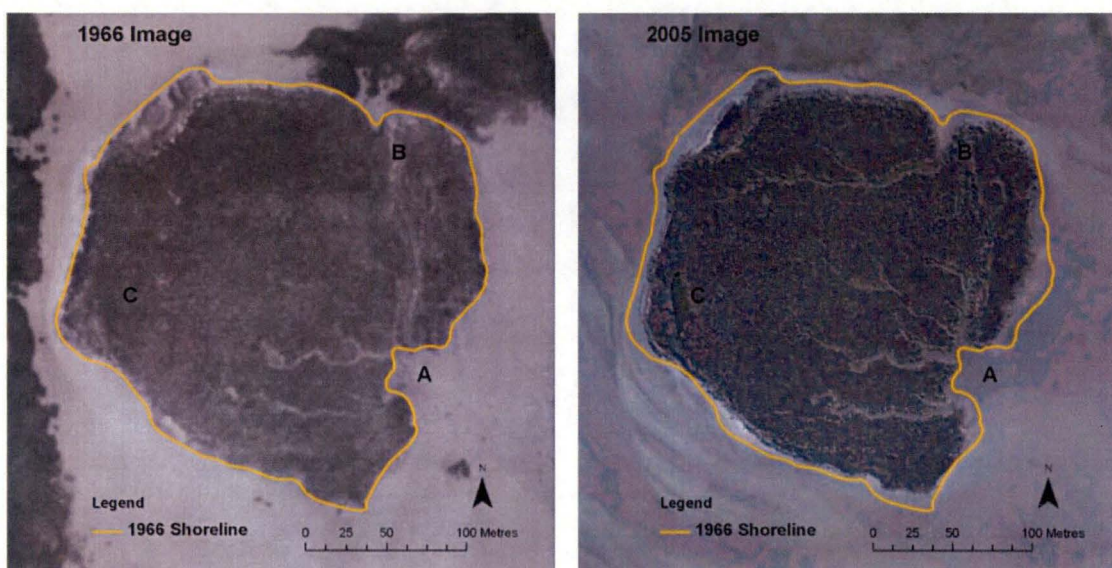


Figure 3.12. Comparison of the 1966 and 2005 aerial photographs indicate: A and B – expansion and widening of tidal channels that are dissecting the marsh; C – increase in the size of the pond.



**Figure 3.13. An example of erosion (undercutting and bank collapse) in tidal channels leading to their lateral expansion (in Barilla Bay).**

#### ***3.2.3.3 Transgression of Shell Ridge/ Sand Sheets***

Another phenomenon noticed during ground and aerial surveys of the saltmarsh shoreline was the migration of shell ridges and sand sheets into the marsh, thus changing the local morphology and hydrology and, hence, the vegetation of the marsh (Chapter 4). A continuous line of shell ridge was observed to have moved onto the marsh in the Iron Creek area where the shoreline had an extremely high wave exposure (Figure 3.14). In the Barilla Bay area (Site 12), a section of the shoreline that is highly exposed to waves has several sand sheets (up to 0.2-0.3 m high in some cases) transgressing several metres into the marsh (Figure 3.15). The tidal creeks on this shoreline have a considerable amount of sand deposited in the channel, starting from its mouth to several metres inwards. Less spectacular transgressions were observed in Site 14 in Barilla Bay, in parts of Ralphs Bay and on one section of Site 20 in Pipe Clay Lagoon. The wave exposure modelling indicates a very high Wind-Fetch Index for all these sections of the saltmarsh shoreline suggesting that the observed shell ridge/sand sheet transgressions were all associated with high wind-wave action.

#### ***3.2.3.4 Patterns of Changes in the Coastal Geomorphology***

Analysis of the areas of advancement and retreat has revealed some general patterns of shoreline geomorphic change. Retreat, primarily caused by erosion of the saltmarsh periphery, was found mostly on shorelines exposed to high wave energy. On the other hand, advancement, due to accretion by sedimentation, was observed primarily in shorelines protected from wind-generated waves. Hence, the geomorphology of saltmarsh shoreline can be said to be directly affected by their degree of 'openness' to the wind. The saltmarshes in the more enclosed parts of the inner estuary or the less exposed areas of the lagoon were most protected. The main areas within the inner estuaries that experienced accretion were the mouths of the Coal River (at Site 3) and Duckhole Rivulet (Site 8). However, it was observed that the tidal channels have widened considerably. Comparison of the aerial photographs indicated that the two main channels in Site 9 at Duckhole Rivulet have widened up to 5 m and 3 m respectively since 1966.

The spit at Iron Creek lengthened due to accretion while the width was reduced due to erosion (Plate 3.1). Overall, there was nearly a one per cent decrease in the vegetated area of the spit with a loss of around 2,300 m<sup>2</sup>. This indicates that processes such as longshore drift can sustain the area of spits and their resident saltmarsh. In Pipe Clay Lagoon, there is a good example of site specific erosion/accretion process as two saltmarshes located on either side of the mouth of the Lagoon have responded differently, one by submergence and the other by seaward advancement (Plate 3.2).





Figure 3.14. Shell ridge over topping into the marsh at Site 16a in the Iron Creek area.



Figure 3.15. Sand sheet transgressing several metres into the marsh at Site 12 in the Barilla Bay area.



3.2.4 Inundation Modelling and Retreat Pathways

Inundation modelling was conducted for all saltmarsh sites with a sea level rise scenario of 1.1 m by the turn of the century. The models indicated that almost all the current extent of saltmarshes have a high probability (>90%) of being under water (Appendix II). Where the landward edge of the marsh is bordered by a gradually rising upland, saltmarshes can be expected to retreat (or transgress) inland. Based on the inundation modelling results, four main scenarios emerged (Figure 3.16). The first scenario applies to eight of the 21 sites, which will be lost due to coastal squeeze (or drowned). This scenario is contingent on an insufficient rate of vertical accretion. All these sites face natural barriers in the forms of steeply rising upland, or artificial barriers, such as roads or levees, and may be unable to cross these barriers. The second scenario is where natural or artificial barriers prevent landward retreat, but, in contrast to scenario one, saltmarshes can still retreat up the river or creek, or move into confined patches between upland barriers. This scenario applies mainly for sites in Duckhole Rivulet and Iron Creek. The third scenario is a special case, where uplands exist as islands surrounded by marsh or intertidal areas or open water, as in the case of Sites 1, 2 and 3. In this scenario, saltmarshes will retreat to higher ground on the islands, but their size will be limited by the (reduced) area of the islands. The fourth scenario is when there is a large area of upland that slopes gently from the marsh. This scenario exists for seven of the 21 sites, with a minimum of one site per study area except in the case of Duckhole Rivulet and Iron Creek.


Sites: 22 19 18 15 14 12 1 	Sites: 3 2 1	Sites: 17 9 8	Sites: 21 16 10 7 6 5 4
Scenario 4	Scenario 3	Scenario 2	Scenario 1
Uplands	Islands	Creeks and Confined Patches	Coastal Squeeze or Submergence

Figure 3.16. Four retreat scenarios for the future of saltmarshes identified from the inundation modelling.

### **3.2.5 Changes in the Landward Extent**

On the landward side, where direct comparison between the historical and recent aerial photographs was possible, there was an estimated amount of 207,883 m<sup>2</sup> loss in the saltmarsh area over the study period. This amounts for about 6% of the total saltmarsh area in 1966. The largest loss was recorded at Site 11 near Shark Point where around 130,326 m<sup>2</sup> was lost due to the construction of a barrier to tidal movement (Plate 3.4). The other major loss was in Barilla Bay where about 45,682 m<sup>2</sup> of saltmarsh area was converted into a freshwater/brackish marsh. Land reclamation on the western side of Orielton Lagoon has reduced saltmarsh area by around 22,219 m<sup>2</sup>, while recent dumping activity was observed at one part of the Lagoon during field and aerial surveys undertaken for this study (in April-May 2009; Plate 3.4).

Other landward edge losses include 7,032 m<sup>2</sup> in Ralphs Bay (associated with road works) and 2,624 m<sup>2</sup> in Site 20 in Pipe Clay Lagoon (to water reservoirs). No major losses were detected in the study sites in the Coal River, Duckhole Rivulet and Iron Creek areas. Other types of losses, where saltmarsh vegetation was replaced by either non halophytic coastal vegetation or bare patches, will be discussed as vegetation changes in Chapter 4.

Three landward edge losses were observed outside the boundaries of the study sites in Coal River, Barilla Bay and Iron Creek areas. Land conversion efforts, similar to that in Site 11, seem to have destroyed a saltmarsh area of approximately 93,583 m<sup>2</sup> near the mouth of the Coal River (Plate 3.5). In the southern end of Barilla Bay, an area of around 7,152 m<sup>2</sup> was claimed for construction purposes. In Iron Creek, the damming of a stream has completely removed an unknown area of saltmarsh (Plate 3.6).

### 3.3 Discussion

The edge erosion and internal dissection of the coastal saltmarshes in the greater Hobart area follows a general pattern of saltmarsh shoreline retreat observed in studies elsewhere. A study of long term erosion rates and patterns in the Greater Thames area of the United Kingdom reported extensive marsh shoreline erosion and internal dissection, with a phase of rapid erosion in the 1970's (van der Wal and Pye, 2004). The authors also noted that the erosion was more severe in outer estuaries and the more exposed parts of inner estuaries as compared to the relatively low erosion in wave-sheltered areas. Similar observations of wind/fetch driven erosion and expansion of tidal channels were reported in a 25 year study of saltmarsh erosion in Essex, UK (Cooper *et al.*, 2001). Similar results were also reported in short term studies which examined retreat rates through field measurements. Day *et al.* (1998) in their study of saltmarshes in Venice Lagoon, Italy, reported that the marsh edge eroded rapidly at a rate of 1.2-2.2 m yr<sup>-1</sup>. They also noted that high energy waves had formed new tidal channels that penetrated into the marsh, lengthening and widening two channels at rates of 0.2-0.6 m yr<sup>-1</sup> and 0.26-0.48 m yr<sup>-1</sup> respectively. Schwimmer (2001) in his study of erosion in Delaware, United States of America reported erosion rates that ranged from 0.20-0.47 m yr<sup>-1</sup> and reported a significant correlation between erosion rates and wave power.

The observed erosion of the seaward shorelines in the saltmarshes in Pitt Water and South Arm ranged from about 0.06 to 0.20 m yr<sup>-1</sup>, with greater rates of retreat in more exposed shorelines than the less exposed shorelines of the inner estuaries which are fetch limited. While these retreat rates differ considerably from other studies, the general pattern of erosion was largely similar. Retreat rates are often specific to a site and depend on the forcing factors involved. High retreat rates are usually recorded when several factors, such as sea level rise, tectonic subsidence and high fetch lengths combine. Phillips (1986) reported a high erosion rate of 3.21 m yr<sup>-1</sup> from aerial photograph interpretation of the Atlantic coast of USA. The erosion was a net effect of sea level rise, coastal submergence, high fetch lengths and anthropogenic effects. On the other hand, a study of the change in extent of two saltmarshes in the east coast of Tasmania reported relatively stable shorelines with a cyclical process of erosion and accretion (Morrison, 2006). However, both the sites were in highly enclosed estuaries and had limited fetch.

Wave energy was observed to be the dominant factor involved in the shoreline changes in the study sites. However, the effect of wave energy is not consistent across the entire saltmarsh shoreline as indicated in this study by the lack of correlation between the erosion outliers and the Wind-Fetch Index (here, a surrogate for wave energy) (see Table 3.3). Incorporation of bathymetry in to the Wind-Fetch Index modelling could assist with explaining these ‘inconsistent’ shorelines (Pepper, 2009). Given that saltmarshes occur on tidal coastlines, another factor that could be considered in analysing the effects of wave exposure on erosion is the duration and variation of the tide (Allen and Pye, 1992). However, the incidence of high energy waves at high tide is a stochastic event and is unlikely to affect the results over 40 years. Other factors that may be involved include longshore drift and high sediment loads (in case of a nearby drain). Hence, while the wave exposure modelling results provided here illustrate a general pattern of erosion and vulnerability of the saltmarsh shoreline to wave exposure, a study of other factors involved could provide more insights into the relationship between erosion and wave exposure (Houshold, pers. comm.; Sharples, pers. comm.).

Some patterns of shoreline erosion and morphological change have been noted from the field surveys. The main characteristics of saltmarsh erosion such as microcliffing, undercutting, slumping and rootmat toppling as described in other saltmarsh erosion studies (e.g. Allen and Pye, 1992; Moreira, 1992; Schwimmer, 2001; Hartig *et al.*, 2002) were noted. Erosion was also found within the tidal creeks which extended and expanded over the study period. Also noted during the study is the transgression of shell ridges and sand sheets over the marsh, especially where the wave exposure is very high. This phenomenon has been suggested to be a result of high energy “storm” waves and has been recorded elsewhere (Allen and Pye, 1992; FitzGerald *et al.*, 2008).

On the landward side, the recorded loss of saltmarsh area between the two study periods was greater than the seaward loss. Of the total area lost, nearly 55% was on the landward side. The loss on the landward side was primarily caused due to land conversion, either by tidal restriction or dumping of fill. These two causes have been reported to be two most prominent human impacts on Australian saltmarshes (Streever, 1997) and were cited as a major threat to Tasmanian saltmarshes as early as 1981



(Kirkpatrick and Glasby, 1981). Yet, tidal exclusion, draining and dumping of fill continue to destroy Tasmanian saltmarshes.

Murray-Wallace and Goede (1991) provided evidence to suggest that Tasmania has experienced tectonic uplift during the middle and late Pleistocene. They also noted that the evidence of the uplift continuing into Holocene was insufficient. Based on this, Pugh *et al.* (2002) calculated the average tectonic uplift rate for Tasmania since middle Pleistocene to be  $0.19 \text{ mm year}^{-1}$ . However, as this rate was calculated over geological time scales, it is uncertain whether it applies at the decadal scale. Hence, they used glacial isostatic adjustment (GIA) models to estimate the land uplift at Port Arthur (around 60 km from the farthest study site) to be  $0.04 \text{ mm yr}^{-1}$ . These data suggest that Tasmania has experienced uplift since the middle Pleistocene, making tectonic subsidence an unlikely cause of saltmarsh retreat.

There is considerable evidence indicating an accelerated sea level rise over the 20<sup>th</sup> century with studies and models suggesting a further acceleration of sea level rise over the 21<sup>st</sup> century (e.g. Church *et al.*, 2008; Gehrels *et al.*, 2008). While the global average rate of sea level rise for the period of 1950 to 2000 was  $1.8 \pm 0.3 \text{ mm year}^{-1}$ , Church *et al.* (2006) estimated a lower rate for the Australian coastline. Their estimate was about  $1.2 \text{ mm year}^{-1}$  for the period of 1920 to 2000. The figure for Tasmania from 1841 to 2002 was estimated to be  $0.8 \pm 0.2 \text{ mm year}^{-1}$  (Hunter *et al.*, 2003). Hence, there is evidence to suggest that sea level rise is a probable reason for the shoreline retreat of the saltmarshes.

Gulev and Hasse (1999, p. 164) noted that “wind waves are considered as an effective indicator of the observed climate changes in atmospheric circulation, because waves integrate surface wind characteristics over space and time.” Hence, changes in global climate can cause variations in the wind-wave climate. Grevemeyer *et al.* (2000) have provided evidence to suggest that the wave climate in the north Atlantic Ocean has increased in energy since 1980 due to an increase in surface temperatures caused by global warming. Change in the wave climate generally results from higher magnitude winds causing wave heights to increase. Such high intensity wind-waves are known to have caused “rapid phases” of shoreline erosion in saltmarshes with retreat rates higher than usually observed in times of low intensity wind-waves (van der Wal and Pye,

2004). The long term records for wind speed obtained from the Hobart Airport show an increase in wind intensity starting from around 1990 with a marked increase from 2002 onwards. This suggests that the wind climate around the study sites might have changed with an increase in strong winds causing high intensity wind-waves. This could have been a major cause of the severe erosion, especially the scarps, observed on many shorelines.

As indicated by Allen and Pye (1992, p. 11) “wind-wave climate has most influence on the horizontal extent of mudflats and saltmarshes, (while) sea level, combined with tidal range (which may be expected to vary with relative sea level), mostly affects their vertical growth.” While results from this study has provided evidence for the loss in the horizontal extent of the saltmarshes caused due to wind waves, the distribution and extent of vertical growth, or accretion, is unknown. Besides wind climate, sea level and tidal range, vertical growth is largely determined by mineral sedimentation, plant productivity and autocompaction (Allen, 2000). With the amount of erosion witnessed in the closed coastal environments near the study sites (Sharples, 2006; pers. obs.), it is possible that sufficient sediment would be available to cause vertical accretion. However, there may be a threshold over which the vertical accretion may not be sufficient to keep pace with an accelerating sea level rise (FitzGerald *et al.*, 2008).

Plant productivity is directly influenced by tidal inundation and the elevation of the saltmarsh surface relative to mean sea level (Morris *et al.*, 2002). An increase in sea level and an accompanying increase in tidal heights may then reduce the productivity of saltmarsh plants. Data on plant productivity is not available for Tasmanian saltmarshes, but can be approximated by standing biomass and condition. Hence, any loss in vegetation cover and condition will affect the productivity of the saltmarsh ecosystem, resulting in a reduced contribution towards sedimentation and vertical growth of the marsh.

Autocompaction depends on the “character and lithological composition of the Holocene stratigraphic sequence and the depth to the Holocene basement” (Allen, 2000, p. 1161). Essentially, autocompaction becomes a significant factor when the substrate is formed mainly of silt and peat, and can be uncommon in sand and gravel substrates. One of the prominent areas of erosion recorded within the study sites was at the eastern

end of Site 12 at Barilla Bay, which had a peat/silt substrate (see Figure 3.8). The erosion observed here was related to the high wave energy but may also have been aided by autocompaction. However, lack of evidence precludes positing any general relationship between autocompaction and shoreline erosion in Tasmanian saltmarshes.

Doody (2008) described three distinct physical states for a saltmarsh: accretional, where there is lateral expansion and vertical accretion resulting in an expanding saltmarsh; semi-stable/ dynamic equilibrium, where although there are phases of both erosion and accretion, there is no overall loss of saltmarsh area; and erosional, where there is a net loss in saltmarsh area and, with sea level rise, the marsh moves landward. Most saltmarshes examined in this study are in the erosional state.

### **3.3.1 Conclusion**

The coastal saltmarshes studied in the Pitt Water and South Arm areas have undergone considerable erosion over the study period. Shoreline erosion was more severe in the open and exposed parts of the estuaries and lagoons and less severe in the inner and more enclosed parts. This was likely due to the effect of wind-generated waves, which have increased in frequency and energy due to sea level rise and a possible change in wind climate. These factors have made previously “protected” shorelines more energetic and susceptible to wave erosion. Any further increase in sea levels will cause further erosion in these “less protected parts of the protected coastlines.” In addition to erosion loss encountered at the saltmarsh periphery, a considerable amount of saltmarsh area has been degraded or destroyed by tidal restriction and dumping fill, with such activities continuing to the present day. Inundation modelling suggests that there is a high probability that, in the absence of insufficient vertical accretion, most saltmarshes will be lost by the turn of the century due to predicted sea level rise of 1.1 m. However, retreat pathways exist for some saltmarshes to migrate landwards or along shore as the sea level rises. The locations of the predicted saltmarsh losses and retreat pathways have been identified in maps in Appendix II.



**Plate 3.1.** Change in the spit structure in the Iron Creek area (Site 16a). The spit has eroded considerably while expanding in length (possibly due to longshore migration) with just 1% decrease in the area.

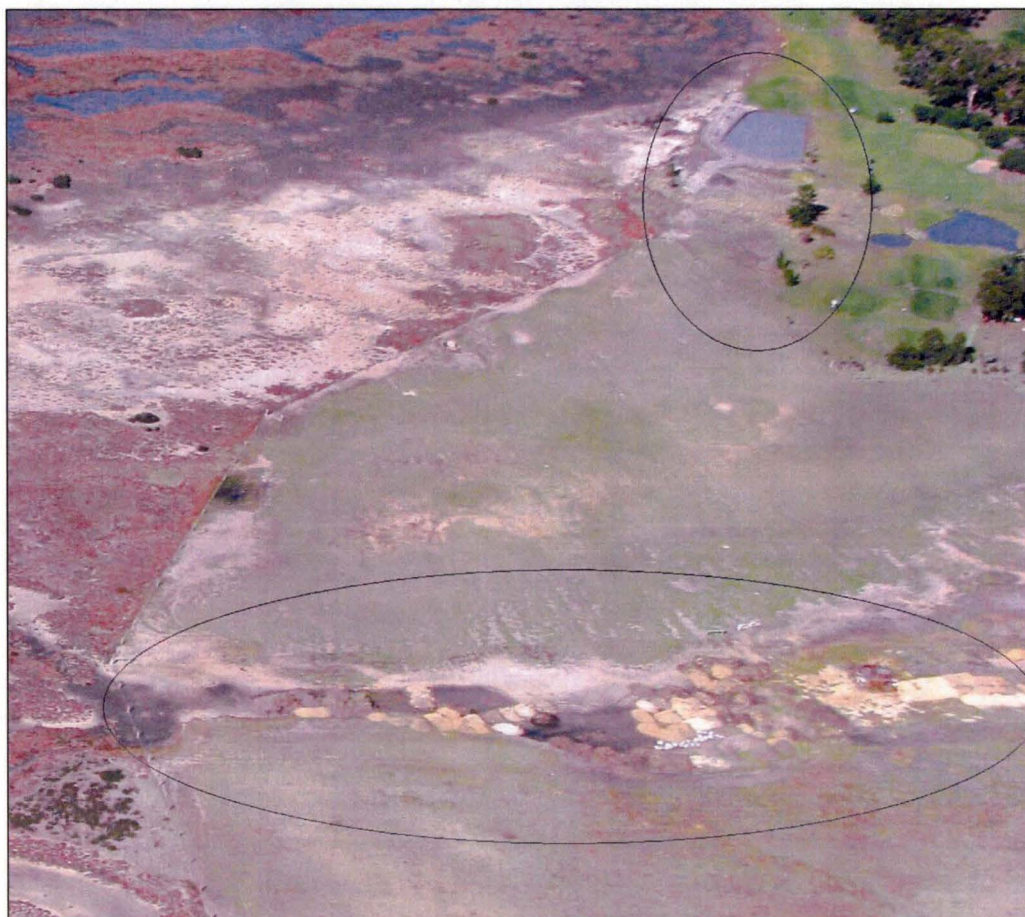


**Plate 3.2.** An example of erosion (coastal squeeze) and accretion, both directed by arrows, at nearby locations in the entrance of the Pipe Clay Lagoon clearly indicating the site specific nature of shoreline change.





**Plate 3.3.** About 130,326 m<sup>2</sup> of tidal saltmarsh area lost following the construction of a tidal barrier near Shark Point in the Pitt Water region.



**Plate 3.4.** Land conversion on the western side of Orielton Lagoon (Site 15). The top right circle shows the marsh converted to golf course and the lower circle shows recent dumping of fill. Also note the marked change in vegetation (from saltmarsh to pasture) along the management boundary.



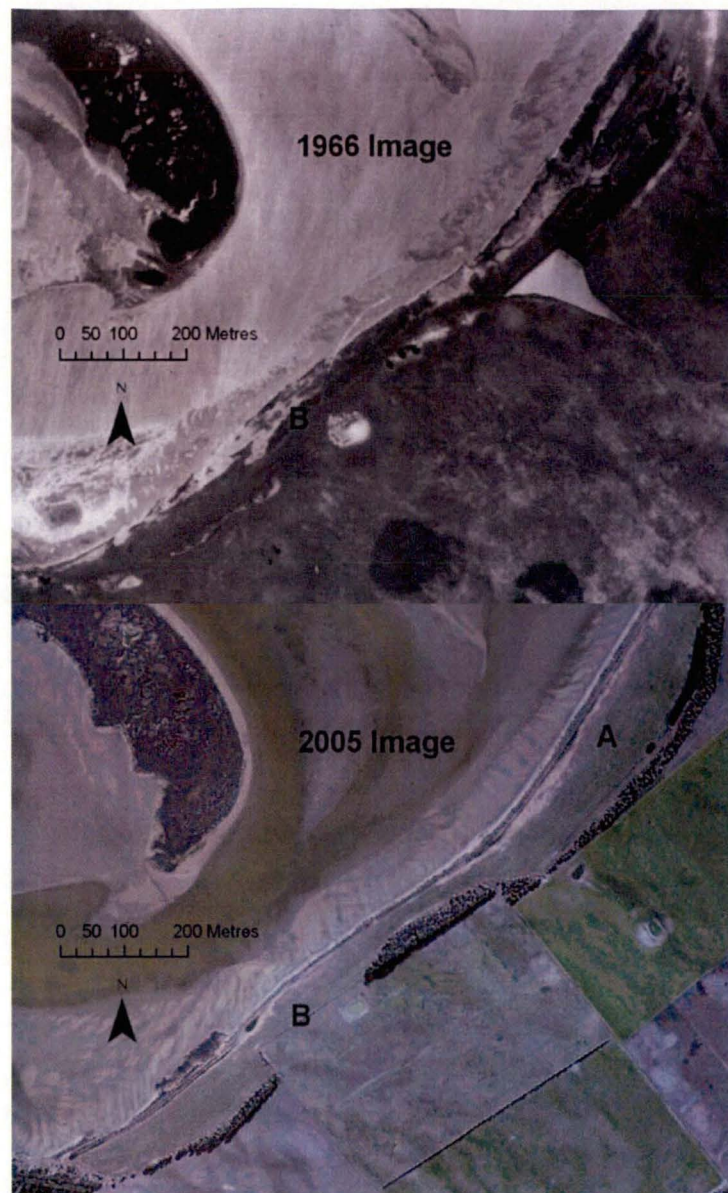


Plate 3.5. The 1966 and 2005 aerial images show (at A and B) the conversion of a saltmarsh area of about 93,953 m<sup>2</sup> in the Coal River area. A closer oblique photograph taken during air survey shows the tidal barrier and the fringing marsh developed seawards from the barrier.



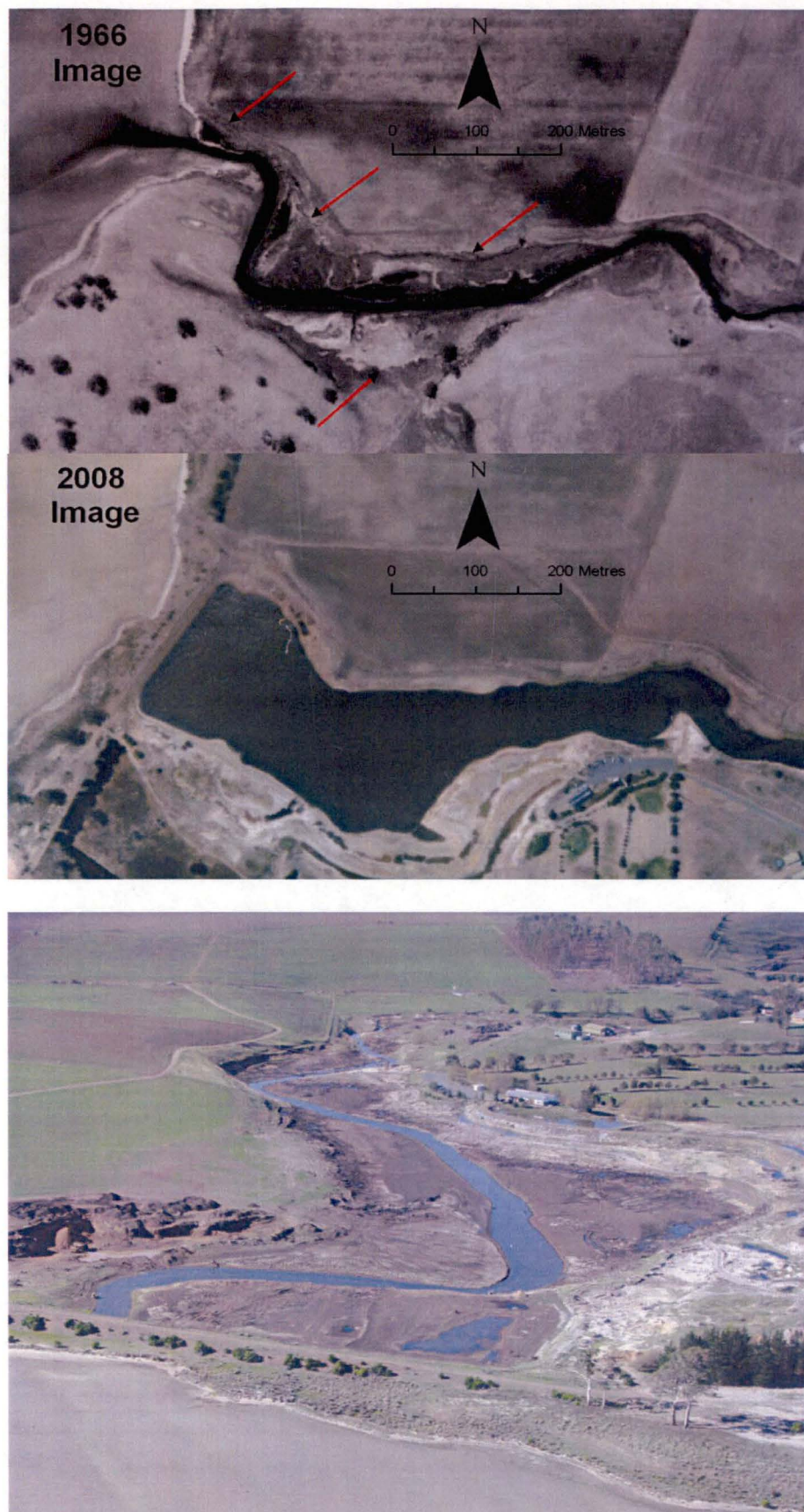


Plate 3.6. The stream at Iron Creek that had been dammed at its mouth. The 1966 image shows the areas of saltmarsh lost due to the damming. The oblique aerial photograph shows extensive continued earthworks operations in the area.

## Chapter 4: Changes in the Vegetation of Saltmarshes

### 4.1 Introduction

Differentiation in the structure and composition of the vegetation of coastal saltmarsh is strongly related to environmental variation (Adam, 1990). Changes in the environment, however subtle or severe, may be reflected in the characteristics of the saltmarsh vegetation. Changes in vegetation will result in changes in the habitat available for the fauna and thereby further affect the ecology and functioning of the saltmarsh ecosystem (Laegdsgaard, 2006). The key environmental variables that have generally been considered to be of importance in understanding vegetation patterns in tidal saltmarsh are: tidal inundation (frequency and duration); local topography (micorelief); substrate (composition and quality); climate (sunlight, temperature, rain and wind); freshwater availability (including groundwater); nutrient supply; and, biotic interaction and succession (Clarke and Hannon, 1969; Ranwell, 1972; Niering and Warren, 1980; Long and Mason, 1983; Bertness and Pennings, 2000; Figure 4.1). Disturbances such as major storms, flotsam and ice-scouring can also affect vegetation patterns in tidal saltmarsh, although the latter does not apply to Tasmania.

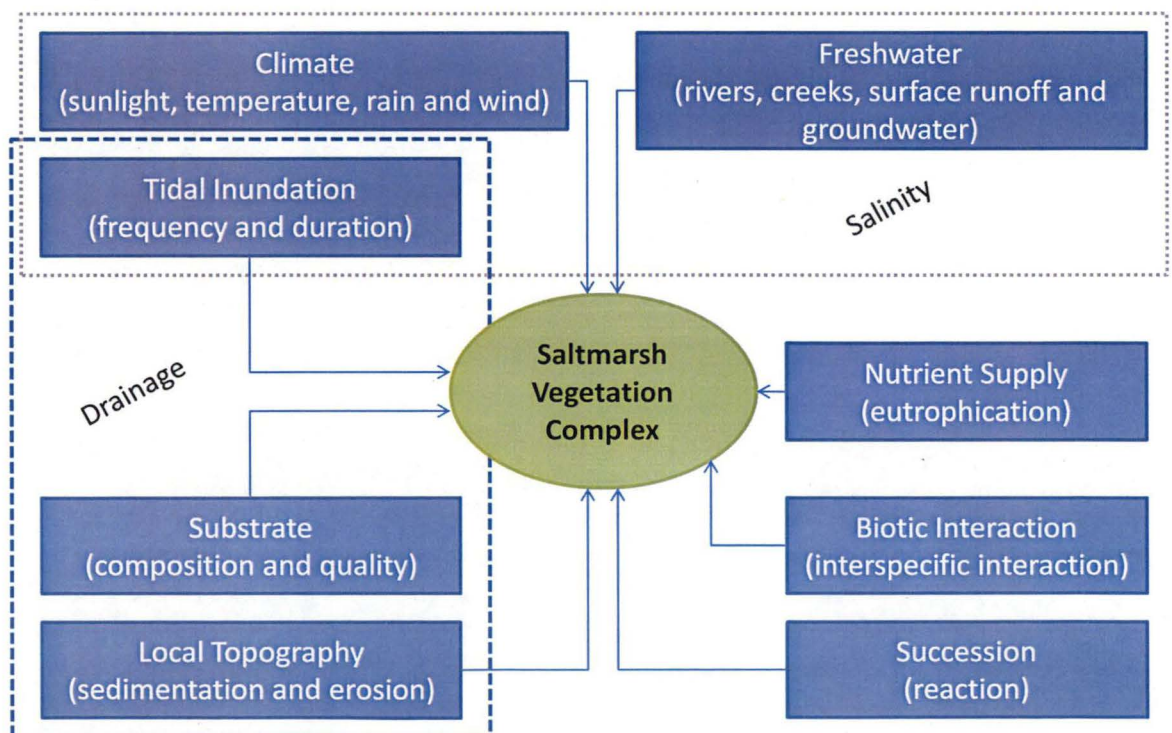


Figure 4.1. Major environmental factors considered to be of importance in determining the structure and composition of the saltmarsh vegetation complex in a given saltmarsh.



The aims of this chapter are to discuss the key processes involved in saltmarsh vegetation change, describe vegetation and cover changes in the saltmarshes of the study area between 1975 and 2008, determine correlates of the changes, and to develop scenarios for future changes.

#### **4.1.1 Key Processes Involved in Saltmarsh Vegetation Change**

Tidal inundation is considered to be the most important factor in the development and constitution of saltmarsh vegetation (Chapman, 1974). Long and Mason (1983, p. 46) observed that “[t]he dominant factor determining the composition of the saltmarsh flora is the ability to withstand seawater inundation.” The relationship between tidal inundation and saltmarsh vegetation is manifest in zonation patterns with plants having different tolerances to tidal inundation occupying different zones in the tidal frame (Davy, 2000).

The two principal effects of tidal inundation on soils, that in turn affect the distributions of saltmarsh plants, are salinity and waterlogging. The degree of salinity varies within the saltmarsh depending upon the periodicity and duration of inundation. The part of the saltmarsh that is not submerged at the mean high water level (emergent marsh or high marsh) can have higher soil salinity than the part of the marsh submerged at mean high water level (submergence marsh or low marsh) as a consequence of evaporation in the high marsh during intertidal periods (Ranwell, 1972). Accumulated salt can be washed away by freshwater runoff. Changes in the periodicity and duration of tidal inundation, solar radiation (increased evaporation) and rainfall can alter the salinity of the high marsh (Bertness and Pennings, 2000).

Waterlogging is another important consequence of tidal inundation and is, to a considerable extent, related to the morphology of the saltmarsh. Waterlogging causes anaerobic conditions in the soil that can stress the root systems and stunt the growth of unadapted plants (Long and Mason, 1983). In places where the submergence period of the high marsh has been observed to increase, vegetation changes have ensued, with waterlogging-adapted lower marsh species replacing the unadapted higher marsh species (Field and Philipp, 2000; Donnelly and Bertness, 2001). The detrimental effect of waterlogging on the growth and reproductive ability of plants has been shown to be more pronounced with high water temperatures (Groenendijk, 1985).

Local topography (or microrelief), while generally reflecting the antecedent landscape, is further shaped by sedimentation (or vertical accretion) and erosion rates and patterns, which are greatly influenced by the tides (Huiskes, 1990). Hence, changes in tidal regimes can result in vertical accretion and/or erosion (both internal and peripheral) and alter the local topography of the marsh. Allen and Pye (1992, p. 11) have suggested that saltmarsh erosion may result in “the enlargement of pans and creeks ... leading to coalescence and appearance of extensive areas of bare mudflat with residual hummocks” and “widespread deterioration of marsh vegetation, leading to generalised scour and surface lowering” having “a profound effect on the vegetation ecology and conservation value of saltmarshes.”

The physical characteristics of the saltmarsh substrate determines the drainage and aeration of the soil and hence affect the “performance and abundance” of saltmarsh plants (Ranwell, 1972, p. 92). Substrate composition of the saltmarsh differs widely, from sand and gravel to clayey or silty soils often mixed with organic material. Sandy and gravelly soils have good drainage and are often occupied by grass-dominated vegetation while clayey and silty soils have poor drainage except adjacent to creeks (Chapman, 1964). The drainage and aeration of the substrate can be considerably affected by tidal restriction and grazing, resulting in changes in vegetation (Roman *et al.*, 1984; de Leeuw *et al.*, 1994; Kuhn and Zedler, 1997; Gouldthorpe, 2000).

Freshwater affects the salinity of the saltmarsh and hence the vegetation composition. The three main sources of freshwater for the saltmarsh are from river flow (for estuarine saltmarshes), groundwater flow, and direct rainfall (Boorman, 2009). Also, irrigation of nearby upland can cause freshwater runoff into saltmarshes. River flow can greatly affect the establishment of saltmarsh communities due to massive discharges of freshwater. In the case of Tasmania, the low energy low rainfall coasts on the east have highly developed saltmarsh communities while massive discharges from the rivers have impeded saltmarsh growth on the low energy high rainfall coasts on the west (Kirkpatrick and Harris, 1999). Freshwater also provides allocthonous inputs of nutrients and sediments. On the other hand, freshwater also act as an agent in transporting pollutants into the saltmarsh, with consequential biotic changes (Boorman, 2009).



Saltmarshes are generally considered to be nitrogen-deficient environments and hence saltmarsh plants have adapted to survive with low levels of nitrogen. An increase in nitrogen levels in the saltmarsh can induce lush and brittle growths in the saltmarsh plants and make them highly susceptible to the physical stress caused by wave action (Ranwell, 1972, pp. 54-55). Furthermore, Levine *et al.* (1998) observed in their study that addition of nitrogen can facilitate the proliferation of certain saltmarsh plants which under normal nitrogen levels are restricted to confined habitats. Hence, nitrogen enrichment, which is usually due to nearby land use practices, can cause interspecific competition, often favouring selected species and alter the community composition of the saltmarsh (Bertness and Pennings, 2000; Bertness *et al.*, 2002; Pennings *et al.*, 2002). In some cases, eutrophication may lead to the loss of perennial plants, these being replaced by fast growing annuals such as blue-green or filamentous algae (Schramm and Nienhuis, 1996). In south-eastern Australia, nutrients are considered to be one of the reasons for the transgression of mangroves into the saltmarsh communities (Saintilan and Williams, 1999).

Bertness (1991) suggested that the lower marsh plant zonation is controlled by relative tolerance to waterlogging while the upper marsh plant zonation is controlled by interspecific interaction. He showed that the competitive ability of a particular species depended on its reproductive faculties, and other growth and morphological features, and that the growth of each upper marsh species increased when isolated from competitors. Interspecific interaction can also be observed in succession, when, after a pioneer saltmarsh species has established on a low lying tidal flat and facilitated sediment accretion, it is displaced by more competitive higher marsh species (Long and Mason, 1983). However, such changes take place over very long periods and few long term studies have attempted to analyse succession in saltmarshes. In one study on long term saltmarsh succession, Roozen and Westhoff (1985, p. 31) failed to identify a relationship between plant zonation and succession and concluded that “[t]he actual differences in the vegetation zonation represent differences in initial environmental conditions and differences in the processes that modified the abiotic environment thereafter” and added that “[o]nly on a small scale, zonation sometimes can be interpreted as succession.”

#### 4.1.2 Environmental Factors Influencing Saltmarsh Vegetation Distribution in Tasmania

The key environmental factors driving the vegetation distribution and zonation (where applicable) in Tasmanian saltmarshes were described by Kirkpatrick and Glasby (1981). The authors, based on their field observations of vegetation distribution, salinity and moisture content in 29 saltmarshes, hypothesised that salinity and drainage were the two main factors that determined saltmarsh vegetation patterns (Figure 4.2). Drainage is essentially a factor that integrates the effects of waterlogging, local topography (marsh elevation) and substrate type. Morrison (2006), investigating factors influencing vegetation zonation in two saltmarshes on the east coast of Tasmania, confirmed that a strong relationship exists between elevation and vegetation zonation. The study suggested that the relationship was stronger in the mid and upper marsh than in the low marsh which was dominated by either *Samolus repens* or *Sarcocornia* spp.. Based on the best available information, a hypothetical model can be suggested that best describes the occurrence of vegetation zones within south east Tasmanian saltmarshes with respect to marsh elevation (Figure 4.3).

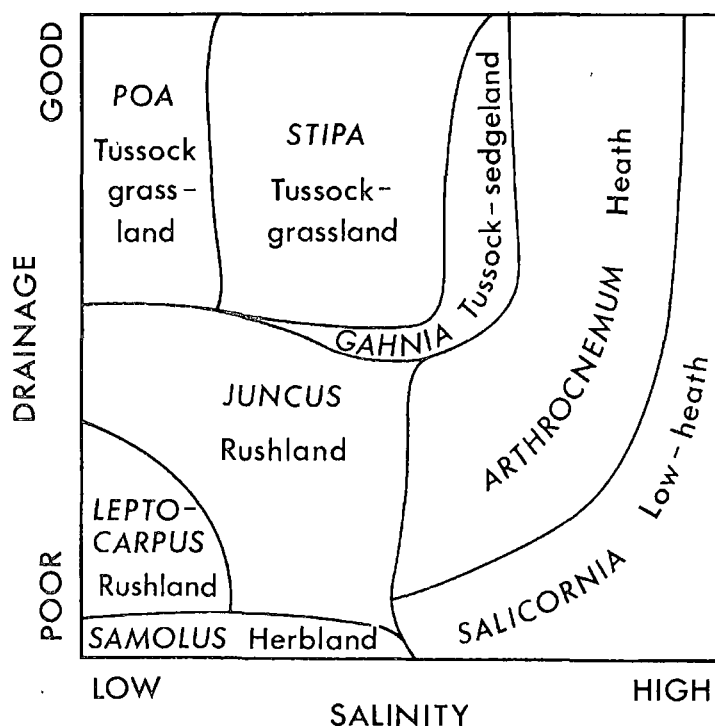


Figure 4.2. Relationship between the major south east Tasmanian saltmarsh communities and the environmental relationships (salinity and drainage) as identified by Kirkpatrick and Glasby (1981, p. 47). Note the name changes: *Salicornia* -> *Sarcocornia*; *Arthrocnemum* -> *Tecticornia*; *Stipa* -> *Austrostipa*; and *Leptocarpus* -> *Apodasmia*.

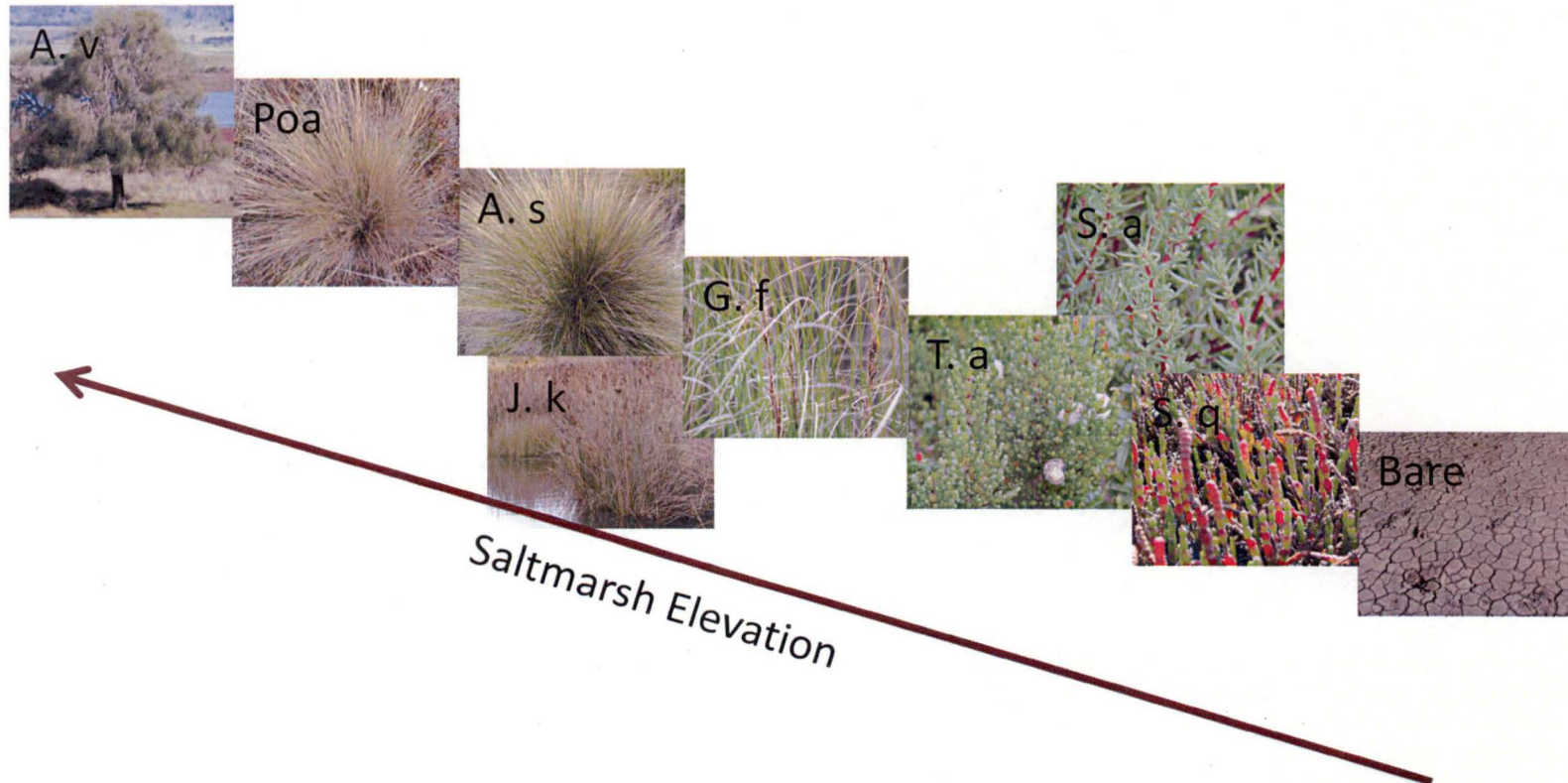


Figure 4.3. A hypothetical model of vegetation zonation for south eastern Tasmanian saltmarshes showing the position of each major dominant vegetation community with respect to saltmarsh elevation (or drainage/waterlogging). Communities include (from right) *Sarcocornia quinqueflora* (S. q), *Suaeda australis* (S. a), *Tecticornia arbuscula* (T. a), *Gahnia filum* (G. f), *Juncus kraussii* (J. k), *Austrostipa stipoides* (A. s), *Poa* spp. (Poa) and terrestrial/coastal vegetation (A. v for *Allocasuarina verticillata*, a dominant species found on the terrestrial edge of south eastern Tasmanian saltmarshes).

## 4.2 Results

### 4.2.1 Saltmarsh Vegetation Mapping

A total of 32 structural components were identified and mapped within the 21 marshes (vegetation maps provided in Appendix III). These components include 30 dominant saltmarsh vegetation communities, bare ground devoid of any vegetation (waterlogged, damp or desiccated) and a common non-saltmarsh vegetation group (abbreviated to NSM) that occurred within or closely associated with saltmarsh vegetation communities. These components together occupied a total area of 3,196,297 m<sup>2</sup>, 89% of which fell in the Pitt Water region (2,839,303 m<sup>2</sup>) and the remaining 11% in South Arm (356,994 m<sup>2</sup>). Overall, eight structural components had a percentage cover of above 1% and were considered to be the major components of the saltmarshes mapped (Figure 4.4).

They were:

1. *Sarcocornia quinqueflora* herbfield – 31.2%.
2. *Tecticornia arbuscula* heath – 29.8%.
3. Bare Ground – 22.4%.
4. *Gahnia filum* tussock sedgeland – 2.9%.
5. *G. filum*/*T. arbuscula* tussock sedgeland/heath – 2.6%.
6. *Juncus kraussii* rushland – 2.2%.
7. *Austrostipa stipoides* tussock grassland – 1.9%.
8. *Suaeda australis* heath – 1.2%.

These plant communities are henceforth named by their dominant/s.

The eight major components accounted for over 94% of the entire saltmarsh area of which more than 60% was occupied by *S. quinqueflora* and *T. arbuscula* (Table 4.1). The former occurred in all the sites while the latter occurred in all sites except in the Iron Creek area (Sites 16 and 17) and were very limited in Sites 19 and 20 in Pipe Clay Lagoon. Bare ground occurred in all sites but was most extensive in Sites 1 and 3 in Coal River, Site 15 in Orielton Lagoon, Sites 12 and 14 in Barilla Bay and in Sites 8 and 9 in Duckhole Rivulet. *G. filum* and *G. filum*/*T. arbuscula* existed in small patches in several sites while the largest patches were recorded in Coal River Sites 1, 2 and 6. *J. kraussii* also occurred in small patches in several sites with the largest patch recorded in Site 3 at Coal River. *A. stipoides* occurred in several small patches scattered within and across sites, especially at the landward border or on ridges on the low or middle marsh.



An exception to this is the largest *A. stipoides* patch recorded in Site 16a in Iron Creek where it, along with *S. quinqueflora*, covered most of the spit. *S. australis* occurred almost exclusively on the seaward margin of the saltmarsh. The largest patch of *S. australis* was recorded in Site 20 in Pipe Clay Lagoon.

The other 24 components were limited in extent and occurred more sporadically with many being specific to a particular site. For example, four of these 24 components – *Apodasmia brownii*/*Poa* spp., *Hemichroa pentandra*, *Atriplex paludosa* and *A. cinerea* – occurred only in Ralphs Bay (Sites 21 and 22). The Ralphs Bay area, comprising only 5.3% of the total study area, recorded the highest number of saltmarsh components (Table 4.1). The non-saltmarsh vegetation group occurred mainly in Site 21 in Ralphs Bay (dominated by *Dodonaea viscosa* and *Allocasuarina verticillata*) and Site 18 in Pipe Clay Lagoon (dominated by *Eucalyptus* spp. and *Leucopogon parviflorus*) while a large patch was recorded in Site 12 in Barilla Bay (dominated by *Lomandra longifolia*).

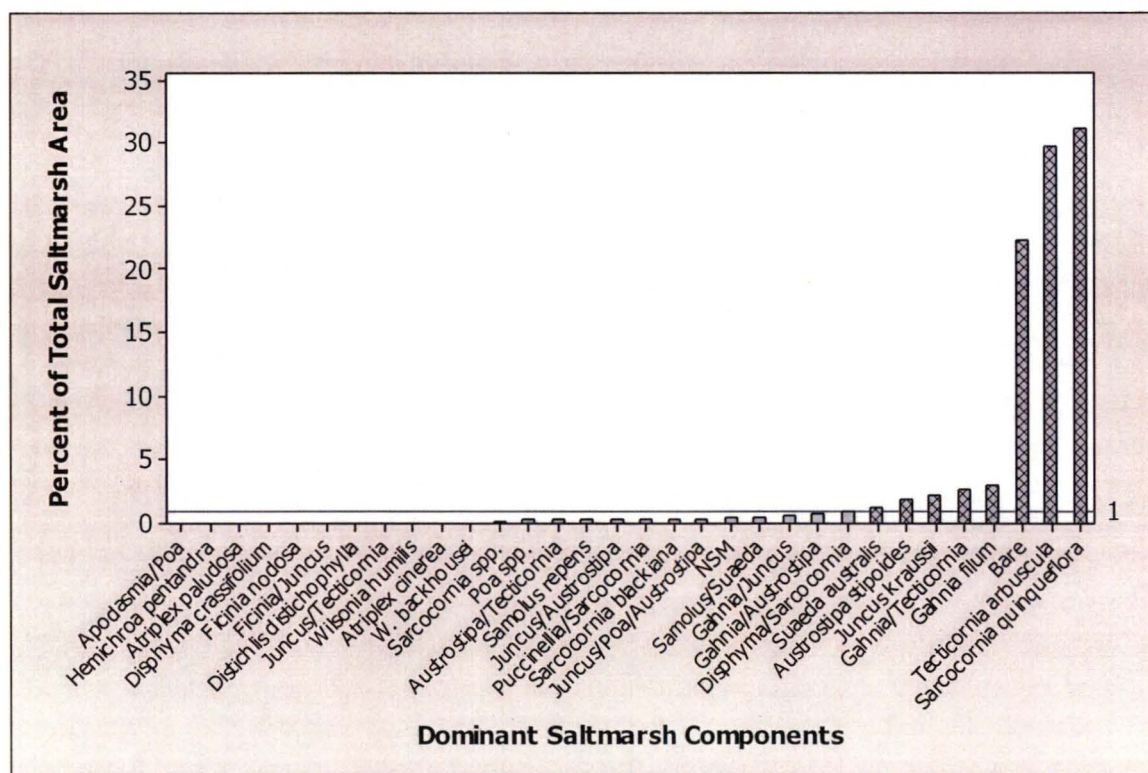


Figure 4.4. The percentage of area occupied by each of the 32 structural components (30 major saltmarsh vegetation types, bare ground and one non-saltmarsh vegetation group) identified and mapped in 2009. The eight components that occupied more than one per cent of the total area are highlighted by crosshatches.

**Table 4.1. The percentage of the total area of marshes, the number of components and the two components with the widest distribution for each discrete marsh.**

Region	Study Area	Site No.	% of Total Area	Component Types Recorded (No.)	First and Second Most Dominant Saltmarsh Components
Pitt Water	Coal River	1	5.8	12	<i>T. arbuscula</i> and <i>S. quinqueflora</i>
		2	3.1	10	<i>T. arbuscula</i> and <i>G. filum</i>
		3	15.5	10	Bare Ground and <i>S. quinqueflora</i>
		4	0.4	11	<i>S. quinqueflora</i> and Bare Ground
		5	9	9	<i>S. quinqueflora</i> and <i>T. arbuscula</i>
		6	7.4	8	<i>T. arbuscula</i> and <i>G. filum</i> / <i>T. arbuscula</i>
		7	3.8	8	<i>T. arbuscula</i> and <i>S. quinqueflora</i>
	Duckhole Rivulet	8	3.8	11	<i>S. quinqueflora</i> and Bare Ground
		9	6.6	15	<i>S. quinqueflora</i> and <i>T. arbuscula</i>
		10	0.4	4	Bare Ground and <i>S. quinqueflora</i>
	Barilla Bay	12	11.8	14	<i>T. arbuscula</i> and <i>S. quinqueflora</i>
		13	1.4	4	<i>T. arbuscula</i> and <i>S. quinqueflora</i>
		14	2.8	12	Bare Ground and <i>T. arbuscula</i>
	Orielton Lagoon	15	13.5	16	<i>S. quinqueflora</i> and Bare Ground
	Iron Creek	16a	1.6	7	<i>S. quinqueflora</i> and <i>A. stipoides</i>
		16b	0.4	6	<i>S. quinqueflora</i> and <i>A. stipoides</i>
		17	1.6	6	<i>S. quinqueflora</i> and <i>G. filum</i> / <i>J. kraussii</i>
South Arm	Pipe Clay Lagoon	18	3.3	14	<i>T. arbuscula</i> and Bare Ground
		19	1.5	9	<i>S. quinqueflora</i> and <i>G. filum</i>
		20	1	9	<i>S. quinqueflora</i> and <i>S. australis</i>
	Ralphs Bay	21	1.7	13	<i>T. arbuscula</i> and <i>S. quinqueflora</i>
		22	3.6	16	<i>T. arbuscula</i> and <i>S. quinqueflora</i>

## 4.2.2 Change Analysis by Area

### 4.2.2.1 Coal River

The two predominant changes in the Coal River area were the loss of *T. arbuscula* and an increase in the area of bare ground. *T. arbuscula* was either converted to bare ground or to *S. quinqueflora*, the latter accounting for more than 80% of the area that changed (91,214 m<sup>2</sup>). *S. quinqueflora* also replaced areas of *G. filum*, *J. kraussii*, *A. stipoides* and *Poa* spp., and transgressed in to the abutting agricultural land at Sites 1 and 3, increasing the saltmarsh area by an estimated amount of 11,480 m<sup>2</sup>. Other communities that were recorded transgressing into the agricultural land were *Poa* spp. and *J. kraussii*.

*T. arbuscula* had expanded into areas previously dominated by *G. filum* and *A. stipoides*. While some patches of *S. quinqueflora* were colonised by *T. arbuscula*, they were more limited in area compared to the reverse transition. Many areas of *G. filum*/*T. arbuscula* and *A. stipoides*/*T. arbuscula* changed to *T. arbuscula*.

There was a substantial increase in bare areas, which resulted mainly from the loss of *J. kraussii*, *Poa* spp., *S. quinqueflora*, *A. stipoides* and *T. arbuscula*. The loss was most severe on Site 3 where tidal restrictions of an area of nearly 500,000 m<sup>2</sup> had resulted in an expansion of bare ground into large areas previously dominated by *S. quinqueflora*. The area was open to grazing by sheep which had visibly impacted the soil and vegetation. The tidally restricted marsh also lost *G. filum* and *S. stipoides* with *J. kraussii* gaining dominance over the grasses and sedges. However, the tidal restriction had favoured the establishment of *T. arbuscula* which was observed to be expanding at the time of field survey.

The dieback of *Poa* spp. had resulted in a big area of bare ground at the back of the marsh in Site 1. While *S. quinqueflora* invaded some of the bare areas, others were still devoid of plant growth. Nearby, a tidal embankment had resulted in transition from *S. quinqueflora*, *P. stricta* and *T. arbuscula* communities to bare ground. The 1975 mapping showed distinct strips of *G. filum* and *S. stipoides* at the back of the marsh and *J. kraussii* at the water's edge all abutting a sharply rising upland. These strips were not recorded in the 2009 suggesting that they had been converted to bare ground.

Close to 1,000 m<sup>2</sup> of bare ground was created in Site 2 in the back of the marsh through the loss of *T. arbuscula*. Similar transitions of *T. arbuscula* to bare ground were also recorded in the back of Sites 5 and 7. Besides *T. arbuscula*, Sites 5 and 7 also had a considerable area that changed from *S. quinqueflora* to bare ground, with the loss approximately at 12,770 m<sup>2</sup> for Site 5 and 2,848 m<sup>2</sup> for Site 7. Two other major changes were an increase in the area dominated by *Sarcocornia blackiana* and *Suaeda australis*. The increase in *S. blackiana* was entirely within Site 1 where it replaced *A. stipoides*, *Poa* spp. and *S. quinqueflora* and was also recorded to have transgressed into agricultural land. The newly formed areas of *S. australis* occurred along the edge of the water, with a notable expanse in Site 1 where it occurred near the mouth of the tidal channel.

#### 4.2.2.2 Duckhole Rivulet

The major changes recorded for Duckhole Rivulet were an increase in the area of vegetation dominated by *S. quinqueflora*, *S. australis*, *S. repens*/*S. australis* and the expansion of bare ground. *S. quinqueflora* replaced *T. arbuscula*, *A. stipoides*, *Puccinellia stricta* and *J. kraussii*. A big expanse of *P. stricta* mapped on the south western end of Site 9 had completely disappeared and the area had changed to *S. quinqueflora* and bare ground, and, to a lesser extent, to *S. blackiana* and *T. arbuscula*. Vegetation dominated by *J. kraussii* found in the upper estuaries (in Site 9) had either converted to *S. quinqueflora* and bare ground or was invaded by *A. stipoides* and *G. filum*. Site 10 had a well defined *J. kraussii* border at the back of the marsh during the 1975 survey, but had subsequently changed to *S. quinqueflora* or bare ground. A similar loss was recorded at Site 9 where well defined *G. filum* and *A. stipoides* patches changed to either *S. quinqueflora* or bare ground. These losses were observed in areas where there existed a steep incline which would have impeded their landward colonisation.

The biggest change was the conversion of *T. arbuscula* to *S. quinqueflora* or *S. australis*. A considerable number of patches dominated by *S. australis* and *S. australis*/*S. repens* were recorded in the low marsh especially at or near the mouth of tidal channels in places formerly occupied by *T. arbuscula*. In another notable change, many sections within the marsh, which were previously vegetated by saltmarsh



perennials, were bare and supported significant amounts of filamentous algae. Most of the algal growth seemed to be relatively recent at the time of field survey and appeared to be moved around the marsh by spring tides (Figure 4.7).

**Table 4.2. Major vegetation changes identified in the Coal River study area.**

Area No.	1975	2009	Site No.
1.	<i>G. filum</i>	<i>S. quinqueflora</i>	1, 2, 3, 5, 6
		<i>T. arbuscula</i>	
		<i>G. filum/ T. arbuscula</i>	
2.	<i>G. filum/ T. arbuscula</i>	<i>S. quinqueflora</i>	1, 2, 6
		<i>T. arbuscula</i>	
3.	<i>J. kraussii</i>	Bare	1, 3
		<i>S. quinqueflora</i>	
4.	<i>J. kraussii/ A. stipoides</i>	<i>S. quinqueflora</i>	3
5.	Not saltmarsh	<i>S. quinqueflora</i>	1, 2, 3
		<i>Poa</i> spp.	
		<i>J. kraussii</i>	
6.	<i>A. stipoides</i>	Bare	1, 2, 3, 7
		<i>S. blackiana</i>	
		<i>S. quinqueflora</i>	
		<i>T. arbuscula</i>	
7.	<i>S. quinqueflora</i>	Bare	1, 2, 3, 4, 5
		<i>S. australis</i>	
		<i>J. kraussii</i>	
		<i>T. arbuscula</i>	
8.	<i>T. arbuscula</i>	Bare	1, 2, 3, 5, 6, 7
		<i>S. quinqueflora</i>	

**Table 4.3. Major vegetation changes identified in the Duckhole Rivulet study area.**

Area No.	1975	2009	Site No.
1.	<i>G. filum</i>	Bare	9
2.	<i>J. kraussii</i>	<i>G. filum</i>	9 and 10
		<i>S. quinqueflora</i>	
		<i>T. arbuscula</i>	
3.	<i>P. stricta</i>	Bare	9
		<i>S. quinqueflora</i>	
4.	<i>A. stipoides</i>	Bare	9 and 10
		<i>S. quinqueflora</i>	
		<i>T. arbuscula</i>	
5.	<i>S. quinqueflora</i>	Bare	9 and 10
		<i>S. australis</i>	
		<i>J. kraussii</i>	
		<i>S. repens/S. australis</i>	
		<i>T. arbuscula</i>	
6.	<i>T. arbuscula</i>	Bare	9 and 10
		<i>S. australis</i>	
		<i>S. quinqueflora</i>	

#### 4.2.2.3 Barilla Bay

Vegetation changes in the Barilla Bay area were dominated by the losses of *A. stipoides* and *T. arbuscula*. The former changed to bare ground, non-saltmarsh communities, *S. blackiana*, *A. stipoides/G. filum* and *T. arbuscula*, while the latter changed to bare ground and *S. quinqueflora* (Table 4.4). The amount of bare ground markedly increased over the period with approximately 25,757 m<sup>2</sup> previously occupied by *A. stipoides*, *T. arbuscula* and *S. quinqueflora* becoming bare. The biggest of these changes was at the back of the north eastern part of the Barilla Bay saltmarsh area where about 14,138 m<sup>2</sup> of bare ground resulted from the die-back of *T. arbuscula* (Figure 4.11). The *T. arbuscula* population near the waterlogged bare patches showed stunted growth and appeared unhealthy (Figure 4.12). Deposits of filamentous algal mats found entangled in the *T. arbuscula* shrubs suggested that they were fully submerged by water. Some of the areas where die-back of *T. arbuscula* occurred were dominated by *S. quinqueflora*.

The change from *T. arbuscula* to *S. quinqueflora* was extensive in the areas of the saltmarsh in Barilla Bay which were exposed to regular tidal inundation. An approximate area of 43,876 m<sup>2</sup> previously dominated by *T. arbuscula* changed to *S. quinqueflora*. The change was particularly striking in the island in Barilla Bay (Site 13) where the area dominated by *S. quinqueflora* increased from around a meagre 664 m<sup>2</sup> to 14,020 m<sup>2</sup> in 2009, occupying more than 30% of the island.

At Site 12 *A. stipoides* was replaced by heathy sedgeland dominated by *Lomandra longifolia*, a non-saltmarsh species. In another instance of change in the same site (western end, near the Cambridge aerodrome), excess freshwater input had replaced *A. stipoides* with *Typha* spp.. Die-back of *A. stipoides* was noticed in several of the ridges in the eastern part of Site 12. Some of the ridges were completely occupied by *T. arbuscula* and some changed to *A. stipoides*/*G. filum*. *S. quinqueflora* changed to bare ground in Site 14 as a possible consequence of increased waterlogging (Figure 4.10). The change from *S. quinqueflora* to *S. australis* dominated vegetation was recorded in two places on the seaward edge of the western side of Site 12.

**Table 4.4. Major vegetation changes identified in the Barilla Bay study area.**

Area No.	1975	2009	Site No.
1.	<i>A. stipoides</i>	Bare	12 and 14
		Not saltmarsh	
		<i>S. blackiana</i>	
		<i>A. stipoides</i> / <i>G. filum</i>	
		<i>T. arbuscula</i>	
2.	<i>T. arbuscula</i>	Bare	12, 13 and 14
		<i>S. quinqueflora</i>	
3.	<i>S. quinqueflora</i>	Bare	12 and 14
		<i>S. australis</i>	

#### 4.2.2.4 Orielton Lagoon

The environmental conditions in Orielton Lagoon have changed drastically over the study period due to the opening of the lagoon to the Pitt Water estuary, facilitating tidal exchange. The 1975 vegetation survey was undertaken when the lagoon had been closed

to tidal flushing and a combination of fire and grazing had obliterated wide areas of *T. arbuscula* (Glasby, 1975). Since 1975 the bare areas have been colonised predominantly by vegetation dominated by *Disphyma crassifolium*, *S. blackiana*, *S. quinqueflora*, *T. arbuscula*, *S. australis* and *A. stipoides*. The pattern of colonisation closely followed marsh elevation. Most of the regularly inundated areas on the low marsh were colonised by *S. quinqueflora*, while the well-drained elevated terraces were dominated by a mixture of *D. crassifolium* and *S. blackiana* (Figure 4.14). The colonisation of *T. arbuscula* was patchy. The two patches of *T. arbuscula* recorded in the 1975 survey had expanded and additionally 20 small patches were recorded, most of which, during field surveys, seemed to be expanding (Figure 4.13).

Other main changes in the lagoon included a loss of *A. stipoides* to *S. quinqueflora* and a shift of *S. quinqueflora* to *S. quinqueflora* and *P. stricta* co-dominated vegetation in three different locations (Table 4.5). The largest area of change from *A. stipoides* to *S. quinqueflora* was recorded in the upper part of the Orielson Rivulet, where there was a general increase in the area of the latter species occurring at the cost of more brackish marsh species *J. kraussii* and *Ficinia nodosa*. While *P. stricta* expanded within the marsh, a big patch of the species that bordered the marsh to its north western side was lost completely. The loss was on a private land (grazing pasture) where, notwithstanding a gentle gradient, the saltmarsh species have not managed to establish (Figure 4.14).

**Table 4.5. Major vegetation changes identified in the Orielson Lagoon study area.**

Area No.	1975	2009	Site No.
1.	Bare	<i>D. crassifolium</i>	15
		<i>S. blackiana</i>	
		<i>S. quinqueflora</i>	
		<i>T. arbuscula</i>	
2.	<i>A. stipoides</i>	<i>S. quinqueflora</i>	15
3.	<i>S. quinqueflora</i>	<i>S. quinqueflora/P. stricta</i>	15



#### 4.2.2.5 Iron Creek

There were six major vegetation changes identified at Iron Creek saltmarsh Sites 16a, 16b and 17 (summarised in Table 4.6). The two major consequences of these changes were an increase in the area of *S. quinqueflora* and bare ground. The area of *S. quinqueflora* expanded considerably, with grasses and sedges giving way, while bare ground was created mainly at the expense of vegetation dominated by *S. quinqueflora*, *G. filum* and *A. stipoides*. Interestingly, the rare species *Wilsonia humilis* took advantage of the niche created by the loss of *A. stipoides* and had successfully colonised the bare ridges at two places (Figure 4.15). The two *W. humilis* patches, at around 900 m<sup>2</sup>, were the two largest patches of the species recorded in all study areas. The extending portion of the spit was primarily being colonised by *S. australis* and successively taken over by *A. stipoides* (Figure 4.16).

The changes at Site 16b were particularly extensive, with major losses of *J. kraussii* and *A. stipoides*/*G. filum*. The amount of bare ground and *S. quinqueflora* increased in extent at low elevations while at higher elevations, previously occupied by *A. stipoides*/*G. filum* *Wilsonia backhousei* was dominant in 2009.

**Table 4.6. Major vegetation changes identified in the Iron Creek study area.**

Area No.	1975	2009	Site No.
1.	<i>J. kraussii</i>	<i>A. stipoides</i>	16b and 17
		<i>S. quinqueflora</i>	
2.	<i>G. filum</i>	Bare	16a
3.	<i>S. quinqueflora</i>	Bare	17
4.	<i>A. stipoides</i>	<i>W. humilis</i>	16a
5.	<i>A. stipoides</i> / <i>G. filum</i>	<i>S. quinqueflora</i>	16b
		<i>W. backhousei</i>	
6.	Bare	<i>S. australis</i>	16a
		<i>A. stipoides</i>	

#### 4.2.2.6 Pipe Clay Lagoon

The major changes in the Pipe Clay Lagoon area were an increase in the extent of *S. australis*, *S. quinqueflora* and *G. filum*. The area dominated by *S. australis* increased by more than 11,600 m<sup>2</sup>. Site 19 did not have any areas dominated by *S. australis* in the 1975 mapping but the species dominated vegetation at the water's edge in 2009 (Figure 4.17). In the biggest change in Site 20, a patch of 5,443 m<sup>2</sup> previously co-dominated by *P. stricta* and *S. quinqueflora* converted to *S. australis*.

*S. quinqueflora* increased in extent mainly by colonising bare ground in Site 19 and displacing *T. arbuscula* within several areas in Site 18. The change within the agricultural land which forms a part of Site 18 was striking, with a considerable reduction in *T. arbuscula* and an expansion of bare areas (Figure 4.18). *G. filum* increased primarily at the cost of *S. stipoides* and *J. kraussii*. The change was observed on the high marsh and in the landward margin where *J. kraussii* and *A. stipoides* used to dominate. Two other commonly observed changes were a change from grasses and sedges to *T. arbuscula* and, in some cases, coastal scrub. In the eastern part of Site 18, about 2,000 m<sup>2</sup> of saltmarsh, previously dominated by *G. filum* and *T. arbuscula*, was replaced by coastal scrub dominated by *Allocasuarina verticillata* and *Eucalyptus* spp..

**Table 4.7. Major vegetation changes identified in the Pipe Clay Lagoon study area.**

Area No.	1975	2009	Site No.
1.	<i>T. arbuscula</i>	<i>S. quinqueflora</i>	18
2.	<i>A. stipoides</i>	<i>S. australis</i>	18, 19 and 20
		<i>G. filum</i>	
		<i>S. quinqueflora</i>	
		<i>T. arbuscula</i>	
3.	<i>G. filum</i>	<i>T. arbuscula</i>	18
4.	<i>J. kraussii</i>	<i>G. filum</i>	18, 19 and 20
5.	<i>S. quinqueflora</i>	<i>S. australis</i>	18, 19 and 20
		<i>G. filum</i>	
		<i>T. arbuscula</i>	
6.	Not saltmarsh	<i>S. australis</i>	18 and 19
7.	Bare	<i>S. australis</i>	18, 19 and 20
		<i>S. quinqueflora</i>	

#### 4.2.2.7 Ralphs Bay

The saltmarsh at Ralphs Bay is a diverse mixture of several vegetation communities and has more changes per hectare than all other saltmarshes in the study area. The two most common changes were from *T. arbuscula* to *S. quinqueflora* and from grass and sedge dominance to *T. arbuscula*. The main net losses were suffered by *Poa* spp. and *A. stipoides*. The former was largely converted to *G. filum*, *A. stipoides* or *T. arbuscula* while the latter changed mainly to *G. filum*, *T. arbuscula*, *Atriplex cinerea* and coastal scrub. Vegetation typically associated with coastal scrub had invaded the saltmarsh occupying an approximate area of 2,280 m<sup>2</sup> while signs of further invasion were evident at the time of field survey (Figure 4.19). Some 'bare' areas within Site 22 had well established seagrass beds. As the 1975 survey did not identify any seagrass areas, it is not possible to conclude that these areas were formed since the 1975 mapping. However, in some areas, *S. quinqueflora* dominated vegetation was stressed by waterlogging while seagrasses were flourishing (Figure 4.20). There was a change in the location of *Samolus repens* dominated vegetation within the marsh at Site 22. While in 1975 it was mapped on the water's edge, it was further inland and more extensive in 2009. Unlike other study areas, the bare areas in Ralphs Bay have actually reduced in their overall area and have been replaced by *T. arbuscula* dominated vegetation.

Table 4.8. Major vegetation changes identified in the Ralphs Bay study area.

Area No.	1975	2009	Site No.
1.	<i>T. arbuscula</i>	<i>S. quinqueflora</i>	21 and 22
2.	<i>A. stipoides</i>	Not saltmarsh	21 and 22
		<i>A. cinerea</i>	
		<i>G. filum</i>	
		<i>T. arbuscula</i>	
3.	<i>G. filum</i>	<i>S. quinqueflora</i>	21 and 22
		<i>T. arbuscula</i>	
4.	<i>Poa</i> spp.	<i>A. stipoides</i>	21 and 22
		<i>G. filum</i>	
5.	<i>S. quinqueflora</i>	<i>S. repens</i>	21 and 22
6.	<i>S. repens</i>	<i>S. quinqueflora</i>	21 and 22
7.	Bare	<i>T. arbuscula</i>	21 and 22





Figure 4.5. Filamentous algae smothering *Tecticornia arbuscula* shrubs in Site 5 at Coal River.



Figure 4.6. A common sight in several saltmarshes – a gradual die-back of *Tecticornia arbuscula* shrubs, with increased waterlogging and algal growth fuelled by nutrient run-offs from neighbouring agricultural land (picture from Site 3).





Figure 4.7. In one of the most striking examples, a *Tecticornia arbuscula* shrub is seen almost completely enwrapped in a thick mat of filamentous algae. Huge growths of algae were observed in both Site 9 and 10 in the Duckhole Rivulet smothering saltmarsh vegetation and obstructing regular plant processes and growth. Photo: K. Komzak.



Figure 4.8. Patterns of “internal marsh erosion” as described by Allen and Pye (1992) visible in Site 8 at Duckhole Rivulet. From bottom to up: coalescence of marsh pools and expansion of bare areas; vegetation hummocks or mud mounds in the middle marsh; and well drained salt flats in the high marsh.





Figure 4.9. Aerial oblique photograph of the north western side of the Barilla Bay area (in Site 12) showing the expanse of bare patches, seen either waterlogged or damp depending on its drainage characteristics. The *Tecticornia arbuscula* plants found close to these bare patches showed stunted growth and appeared unhealthy (see Figure 4.12).



Figure 4.10. Aerial oblique photograph of Site 14 showing extensive areas of bare waterlogged areas which have a considerable algal growth.





Figure 4.11. Die-back of *Tecticornia arbuscula* at the back of the saltmarsh at Site 12 has increased the amount of bare ground considerably.



Figure 4.12. Another location in Site 12 showing a different process involved in the die back of *Tecticornia arbuscula* (and subsequent bare ground creation). Here, the artificial channel created (at the bottom left corner) brings in nutrient rich effluents from the nearby businesses favouring algal growth, which is seen deposited on *T. arbuscula* shrubs. The surviving *T. arbuscula* shrubs close to the bare ground, unlike the healthy shrubs located near the bare ground in Figure 4.11, were unhealthy, had stunted growth with algae sticking on to several individuals (possibly transported by the high ebb tides).





Figure 4.13. *Tecticornia arbuscula* colonising bare patches is an example of the succession process taking underway in Orielton Lagoon (Site 15).

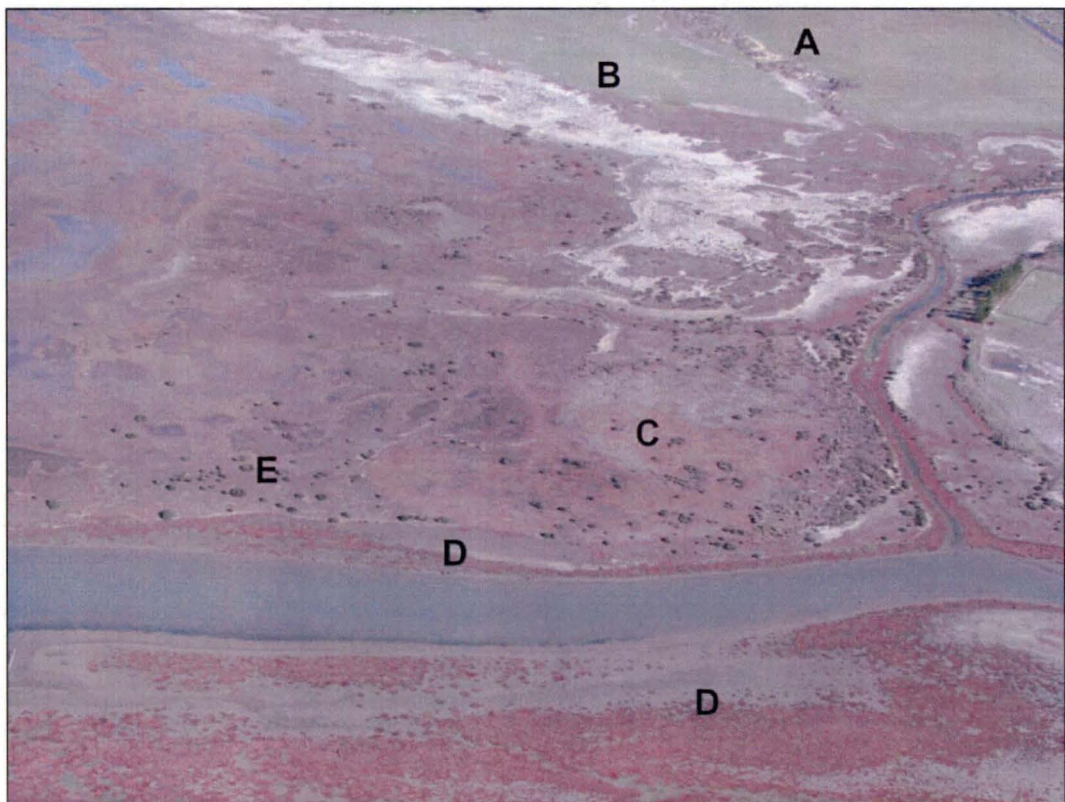


Figure 4.14. Aerial oblique photograph showing the northern part of Orielton Lagoon saltmarsh area (Site 15). A - Dumping in the nearby private land, also see Plate 3.4; B - Striking difference in vegetation across the management boundary, also see Plate 3.4; C - *Disphyma crassifolium* and *Sarcocornia blackiana* dominated terrace; D - *S. quinqueflora* dominated low marsh; E - *Tecticornia arbuscula* colonising and expanding.





Figure 4.15. *Wilsonia humilis*, the grey coloured species, seen well established on the elevated ridge previously occupied by *Austrostipa stipoides* (in Site 16a).

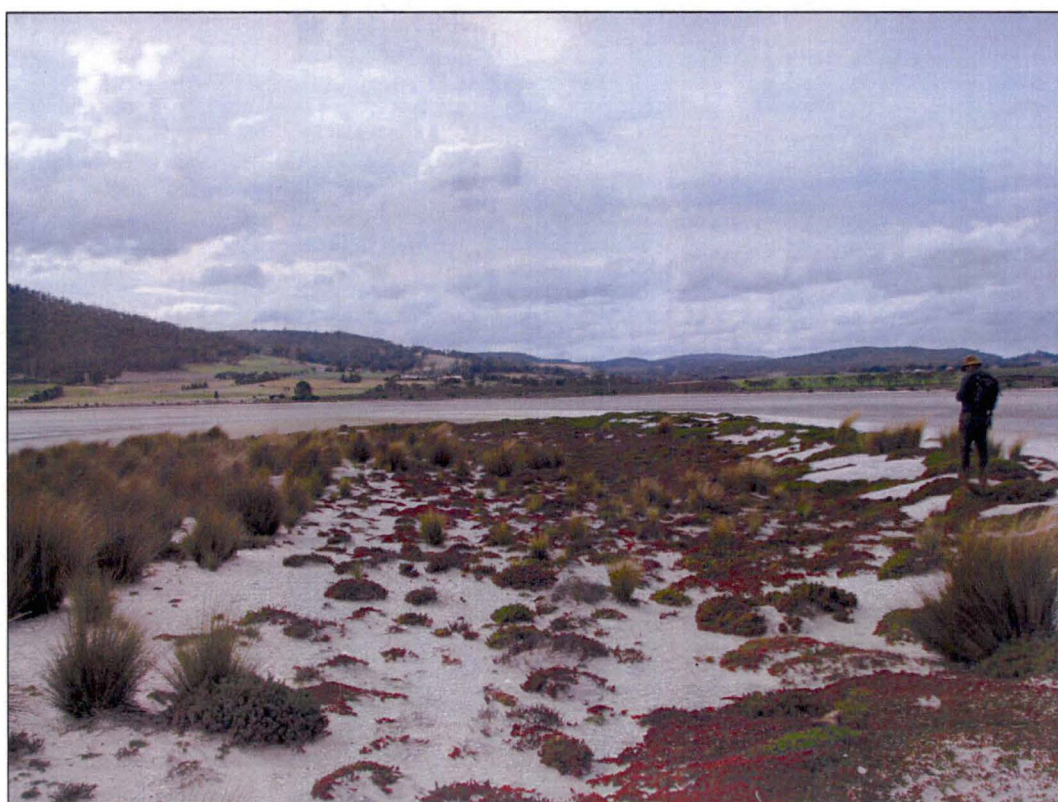


Figure 4.16. Vegetation on the expanding ridge (in Site 16a) comprising of *Suaeda australis* dominated community colonising the newly formed ground, with *Austrostipa stipoides* seen moving into the well drained sections of the pioneer saltmarsh.





Figure 4.17. A continuous stretch of *Suaeda australis* dominated community seen on the shell ridges near the water's edge in the middle and southern parts of Site 19 in Pipe Clay Lagoon. Also seen is *Austrostipa stipoides* taking over *S. australis* as the dominant vegetation of the ridge.



Figure 4.18. Oblique aerial photograph of the north western side of Site 18 shows a sharp variation in the saltmarsh vegetation diversity (and presence of *Tecticornia arbuscula*) across the management boundary (fence). Also seen is the huge expanse of bare ground with the pioneer species transgressing into the farmland. Photo: R. Mount.





Figure 4.19. *Allocasuarina verticillata* seen invading the *Gahnia filum* sedgeland in Site 21 at Ralphs Bay. Also seen in the background is the coastal scrub (with *Dodonaea viscosa*) that had displaced *Austrostipa stipoides* tussock grassland.



Figure 4.20. Increased inundation of seawater seems to be facilitating the establishment of seagrasses in areas dominated by *Sarcocornia quinqueflora* in Site 22 at Ralphs Bay.

### 4.2.3 Major Trends in Vegetation Change

While multifarious vegetation changes were recorded within and across the study areas/sites, some changes were more frequent and extensive than others. Forty-one such “high magnitude” changes were identified based on the number of instances a particular type of change had occurred and the area affected (Table 4.9). These changes involved 13 different saltmarsh plant communities, bare ground and a general non-saltmarsh plant community. Of these, six saltmarsh plant communities – *A. stipoides*, *G. filum*, *J. kraussii*, *S. australis*, *S. quinqueflora* and *T. arbuscula* – were involved in a high percentage of the recorded changes.

*A. stipoides* changed mainly to *T. arbuscula*, *S. quinqueflora* and *G. filum*. Several instances of *A. stipoides* becoming bare ground were recorded, this occurring in the landward margins of the marsh and on ridges both in the mid/high marsh and on the seaward edges. *G. filum* changed primarily to *T. arbuscula* and to a lesser extent to *S. quinqueflora*. The dominant change for *J. kraussii* was to *S. quinqueflora*. About 174,000 m<sup>2</sup> of *S. quinqueflora* was converted into bare ground. Nearly 70% of this area fell within Site 3 which was tidally restricted after 1975. Several changes of *S. quinqueflora* to *S. australis* or *T. arbuscula* were recorded, with the former occurring mostly on the seaward margins of the saltmarshes and creek mouths. More than 50% of the area that changed from *S. quinqueflora* to *T. arbuscula* was within the tidally restricted section of Site 3. *T. arbuscula* changed to *S. quinqueflora* in the biggest vegetation change recorded in the present study. Ninety-six instances of this change were recorded with the total area of change being 184,833 m<sup>2</sup>. A large area of *T. arbuscula* changed to bare ground, with nearly 50% of this area occurring within Site 12 at Barilla Bay.

In other notable changes, *Puccinellia stricta* and *Poa* spp. changed to either *S. quinqueflora* or bare ground, with the latter also changing to *A. stipoides* or *G. filum*. A sizeable area (32,782 m<sup>2</sup>) dominated by *G. filum*/*T. arbuscula* changed to *T. arbuscula* dominance. Many non-saltmarsh communities, usually comprised of coastal scrub, were recorded in the present study. They have mainly replaced *A. stipoides*, *J. kraussii* and *G. filum*, with the changes occurring mostly at the terrestrial fringes of the marsh. A substantial amount of bare ground (161,945 m<sup>2</sup>) became vegetated, mainly with *S.*



*quinqueflora*, *T. arbuscula* or *S. australis*. Of this, nearly 90% converted to *S. quinqueflora*, with more than 90% of this area falling within Orielson Lagoon. Almost 80% of the transition from bare ground to *T. arbuscula* also occurred within Orielson Lagoon.

Of the 41 high magnitude changes, 14 occurred in more than 10 instances (highlighted in Table 4.9; Figure 4.21). These changes occurred across several sites and can be considered to be the strongest “vegetation shifts” recorded in the present study. The biggest shift in terms of both number of occurrences and area was from *T. arbuscula* to *S. quinqueflora*. The second biggest shift was from *S. quinqueflora* to bare ground. There was also a considerable amount of change from bare ground to *S. quinqueflora*. Both these changes were concentrated in Site 3 in Coal River and Site 15 in Orielson Lagoon. Two other major vegetation changes – *S. quinqueflora* to *T. arbuscula* and Bare to *T. arbuscula* – were also within Sites 3 and 15. Changes from grasses and sedges to *T. arbuscula* dominance were widely recorded. *A. stipoides* changed widely to either bare ground, *G. filum* or *S. quinqueflora*. *G. filum* and *J. kraussii* changed to *S. quinqueflora*, while several patches of *S. australis* replaced *S. quinqueflora*.

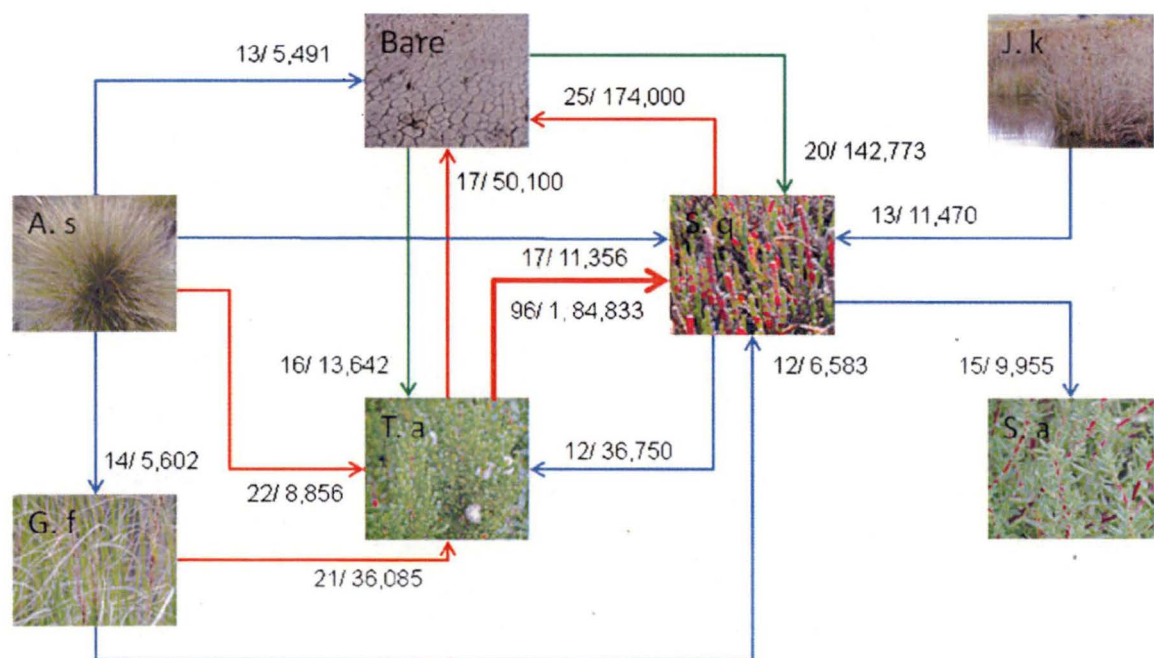


Figure 4.21. Vegetation changes observed between the six major dominance communities and bare ground. Communities are *Austrostipa stipoides* (A. s), *Gahnia filum* (G. f), *Juncus kraussii* (J. k), *Sarcocornia quinqueflora* (S. q), *Suaeda australis* (S. a) and *Tecticornia arbuscula* (T. a). Red lines indicate large changes and the thick red line, the largest change. Green lines indicate secondary succession/landward transgression. The numbers associated with the changes are the number of instances a particular change has been recorded and the total area (m²) of change.

Table 4.9. Magnitude of change in dominant communities from 1975 to 2009 measured in terms of both number of instances of change (polygons/patches) and the area (in m<sup>2</sup>) of change. \* is not applicable and # not significant. >10 instances of change have been highlighted.

Columns – 1975 Rows – 2009	A. stipoides	Bare	G. filum	G. filum/ T. arbuscula	J. kraussii	Not saltmarsh	Poa spp.	P. stricta	S. quinquef- lora	T. arbuscula
A. stipoides	*	#	3/ 2, 837	#	3/ 1, 767	#	4/ 2, 205	#	#	#
Bare	13/ 5, 491	*	7/ 4, 098	#	3/ 4, 176	#	5/ 5, 544	3/ 13, 536	25/ 1, 74,000	17/ 50, 100
G. filum	14/ 5, 602	#	*	#	5/ 5, 791	#	3/ 2, 476	#	4/ 3, 532	#
G. filum/ S. stipoides	6/ 3, 478	#	#	#	#	#	#	#	#	#
G. filum/ T. arbuscula	#	#	5/ 3, 027	*	#	#	#	#	#	#
J. kraussii	#	#	#	#	*	#	#	#	4/ 8, 251	#
Not saltmarsh	8/ 8, 100	#	3/ 1, 736	#	3/ 4, 228	*	#	#	#	#
S. repens	#	#	#	#	#	#	#	#	4/ 2, 673	3/ 1, 196
S. repens/ S. australis	#	#	#	#	#	#	#	#	4/ 4, 362	#
S. blackiana	8/ 2, 445	#	#	#	#	#	#	#	#	#
S. quinqueflora	17/ 11, 356	20/ 1, 42, 773	12/ 6, 583	#	13/ 11, 470	7/ 11, 480	6/ 11, 933	4/ 10, 100	*	96/ 1, 84, 833
S. australis	#	4/ 5, 550	#	#	#	#	#	#	15/ 9, 955	6/ 4, 968
T. arbuscula	22/ 8, 856	16/ 13, 642	21/ 36, 085	4/ 32, 782	4/ 1, 900	#	#	#	12/ 36, 750	*
Instances of Loss:Gain	88:10	40:73	51:26	4:0	31:4	7:14	18:0	7:0	68:175	122:79



#### 4.2.3.1 Percentage Changes in Major Dominance Groups

Glasby (1975) identified six major dominance groups within the 34 dominance groups (mapping units) he employed in mapping the saltmarshes in the study sites. They were: *S. quinqueflora*, *G. filum*/*T. arbuscula*, *T. arbuscula*, *J. kraussii*, *A. stipoides* and *G. filum*. These major dominance groups accounted for 91% of the total saltmarsh area, including some sites in the Derwent estuary not covered above. A comparison of the current percentage cover of each of these major dominance groups between the 1975 and 2009 mapping efforts indicate that all groups have changed in their extent. *S. quinqueflora* considerably increased in proportion from about 32% to 37.8% of the area. On the other hand, *T. arbuscula*, which had the highest percentage cover in 1975 (at 39%), reduced to 36%. *A. stipoides* decreased from 6 to 2.4% in 2009 indicating that more than 50% of saltmarsh area dominated by *A. stipoides* has been lost. *J. kraussii* dominance also reduced from 7.5% to 6.2%. However, both *G. filum*/*T. arbuscula* and *G. filum* gained in percentage cover (Table 4.10).

**Table 4.10. Percentage change in vegetation cover of the six major dominance groups recorded in 1975, with major changes highlighted. \* Figures have been provided both by including and excluding some Derwent River saltmarshes as they were a part of the 1975 study and the percentage values for 1975 includes these additional saltmarshes, which were also surveyed in the present study, but not reported above.**

Major Dominance Groups	% Cover in 1975 including Derwent	Cover in 2009 including Derwent*	
		% Cover	Change
<i>S. quinqueflora</i>	32	37.8	+ 5.8
<i>G. filum</i> / <i>T. arbuscula</i>	3	3.2	+ 0.2
<i>T. arbuscula</i>	39	36.1	- 2.9
<i>J. kraussii</i>	7.5	6.2	- 1.3
<i>A. stipoides</i>	6	2.4	- 3.6
<i>G. filum</i>	3.5	4.1	+ 0.6

## 4.2.4 Environmental Changes in the Study Areas

### 4.2.4.1 Climatic Changes and Sea Level Rise

Rainfall and temperature have changed considerably over the last five decades (1959-2008) with the former showing a decreasing trend and the latter an increasing one (Figure 4.22; Figure 4.23). The mean annual precipitation in the 1960's and the 1970's was in the range of 500 to 600 mm, but dropped to below 500 mm over the last three decades. Rainfall was particularly low over the last decade with an average annual precipitation of about 430.4 mm compared to the 50 year average of 497.1 mm (Bureau of Meteorology, 2009). This is in sharp contrast with the high average annual precipitation of 583.3 mm recorded for the 10 year period before the previous survey in 1975. The highest annual rainfall over the last five decades was recorded in 1975 (735.4 mm) while the two lowest records were for the year 2006 (297.2 mm) and 2008 (329.2 mm). In other words, 2008 received less than half the amount of rainfall recorded for 1975. The 50 year weather records indicate that this variation is no anomaly but a long term trend of decreasing rainfall over the study area. The mean number of rain days has reduced from 151 days in 1975 to 140 days in 2009 (Glasby, 1975; Bureau of Meteorology, 2009).

The temperature has followed the opposite trend as precipitation over the last five decades. The mean daily maximum temperature has increased by almost one degree since 1959. The mean daily maximum temperature for 1999-2008 was 18 °C. This is 0.5 °C above the overall average and 0.67 °C higher than the period between 1966 and 1975. The last decade also had some of the warmest years on record with mean daily maximum temperatures of above 18 °C for five out of ten years, and 2007 recording the highest of 18.4 °C. The mean daily minimum temperature also increased after 1959, with the period between 1999 and 2008 recording an average of 8.53 °C as compared to 7.86 °C recorded between 1966 and 1975.

Sea level has risen around south eastern Tasmania by about 1.2 mm year<sup>-1</sup> for the period between 1920 and 2000 (Hunter *et al.*, 2003). This figure has been calculated by discounting the effects of tectonic movements. The sea level rise is expected to have increased the tidal prism (volume of tidal water) of the enclosed waterbody, namely estuary and lagoon, and consequently increasing the duration and frequency of tidal



inundation (Cooper *et al.*, 2001). Climate change and sea level rise will also affect the wind and wave climate which may affect the saltmarsh (Allen and Pye, 1992; van der Wal and Pye, 2004).

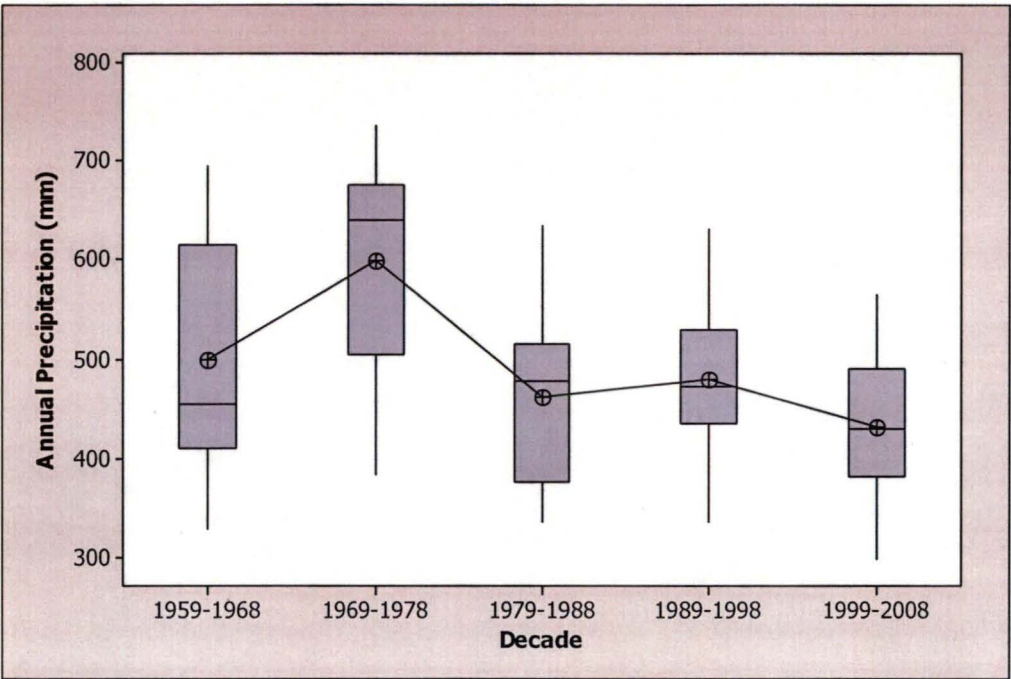


Figure 4.22. Box plot showing the decreasing trend in annual precipitation recorded at Hobart Airport from 1959 to 2008. Also noticeable is the reduced intra-decadal variability in precipitation over the time. The middle line represent the median value, and the circle with a cross the mean. Whiskers extend to the maximum and minimum data points within 1.5 box heights. Observations that lie beyond the range of the whiskers (outliers) are represented by stars.

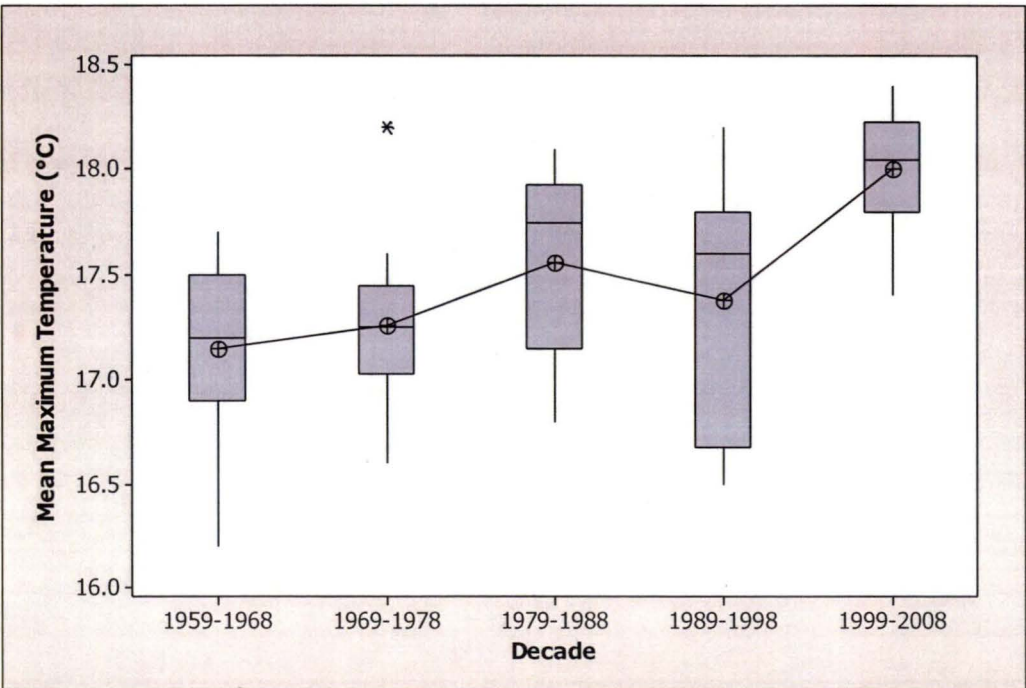


Figure 4.23. Box Plot showing a rising trend in the mean daily maximum temperature data recorded at Hobart Airport from 1959 to 2008. Refer to Figure 4.22 for information on symbols used in the box plot.

#### ***4.2.4.2 Freshwater Availability and Nutrient Loading***

All sites in the Pitt Water region are within the Coal River catchment which is one of the driest catchments in Tasmania, with the water flow dependent on a highly variable rainfall in the region and groundwater inputs (Davies, 2002; DPIWE, 2003a). The natural flow regime is a very high winter flow and much reduced summer flow. For Coal River, this flow regime has been reversed to a high summer flow and low winter flow since the installation of the Craighourne Dam. Furthermore, the amount of water flowing through the river has decreased considerably over the years, with water being diverted for South East Irrigation Scheme and other stock and domestic uses (DPIWE, 2003a). There are a considerable number of boreholes around the study sites in Pitt Water area (Bacon and Latinovic, 2003), but the amount of ground water extracted is not known.

There are several creeks that drain into the Pitt Water estuary and have saltmarshes developed at their mouths. A large number of these creeks have been dammed for stock water supply and other uses (pers. obs. from aerial photographs and field surveys; Davies, 2002). Some creeks have been dammed right at their mouths, eliminating saltmarshes and excluding freshwater flows (Chapter 3). The majority of water resources in the Coal River catchment have been allocated to irrigation (Resource Planning and Development Commission, 2003). Irrigation can considerably increase surface run-off and carry nutrients (mainly nitrogen, phosphorus and potassium) into the saltmarsh and other intertidal environments (Chapter 2). Davies (2001, p. 11) referred to the Coal River Irrigation Scheme as a “cogent example” in Tasmania of mismanagement of water resources leading to environmental issues such as blue-green algal blooms, salinity and decreasing river health.

In Pipe Clay Lagoon and Ralphs Bay, there are a number of storm water drains that run into saltmarshes. The saltmarsh vegetation has changed in some places due to the freshwater inputs from the drains (Figure 4.24). The drains also bring in nutrients and induce algal growth in the saltmarsh (Figure 4.25).





Figure 4.24. Source of freshwater (from a drain nearby Site 22 in Ralphs Bay) has favoured the establishment of a *Samolus repens* zone between the *Sarcocornia quinqueflora* (left) and *Tecticornia arbuscula* (right) zones.



Figure 4.25. A drain in Site 22 in Ralphs Bay bringing in freshwater and nutrients (evidenced by the presence of filamentous algae) into the saltmarsh.



### 4.3 Discussion

The vegetation dieback and the changes in vegetation dominance observed in the south east Tasmanian saltmarshes mostly follow a general pattern of change observed in saltmarshes elsewhere in Australia and in several other parts of the world. Vanderzee (1988), in one of the earliest accounts of saltmarsh vegetation change due to sea level rise in Australia, detailed the process of change in vegetation communities with the aid of aerial photographs (from 1941 and 1984) and direct field observations. He studied saltmarshes at Corner Inlet, Victoria, which had similar saltmarsh species and community structure as the rest of south-eastern Australia (except for the absence of a seaward mangrove zone in Tasmania). The main findings from the study which can be directly related to the results reported here for Tasmanian saltmarshes are a marked increase in bare ground resulting from the dieback of *T. arbuscula* and the dominance of *S. quinqueflora* in areas previously dominated by *T. arbuscula*. He also added that the dieback of *T. arbuscula* was more prominent on the seaward edges and near the marsh ponds.

Similar results have been reported for some North American saltmarshes. Donnelly and Bertness (2001) reported that the high marsh communities in New England were being stressed by increased flooding caused by sea level rise and were gradually making way for the landward migration of low marsh communities. They further stated that, with current rate of sea level rise, the present New England saltmarshes will completely be dominated by low marsh species over the next century. In Delaware Estuary, New Jersey, Field and Philipp (2000) reported that the high marsh vegetation, especially at the lower parts of the tidal creeks, have been replaced by low marsh vegetation. Based on the changes observed, they suggested that “[v]egetation provides an indication of hydroperiod so that change in vegetation can signal change in tidal duration and depth as well as change in the relative elevation of the marsh plain.” Hence, it is probable that the dieback of *T. arbuscula* and the increase in *S. quinqueflora* dominance widely seen in the study sites was caused by an increase in hydroperiod (waterlogging) affected by changes in tidal flooding frequency and duration due to sea level rise.

While increased tidal inundation is known to affect the high marsh species and favour low marsh species, the opposite can be said to be true if the tide is restricted (with all

other factors remaining constant). Some evidence for this reverse-change hypothesis was found at the tidally restricted sections of Site 3 where patches previously dominated by *S. quinqueflora* changed to *T. arbuscula* which also seemed to be expanding further. This follows the results reported by Esselink *et al.* (2002, p. 27) that “[a] decline in flooding frequency in salt marshes normally leads to an increase of middle- and high-marsh species and a decrease of pioneer and low-marsh species.” However, in the case of Site 3, the newly formed *T. arbuscula* patches occur in a basin which can hold a significant amount of freshwater during heavy rain. Freshwater inundation can lead to the dieback of *T. arbuscula* plants. So, while the current anthropogenic influence on tidal flows might have favoured *T. arbuscula*, when environmental factors change, the vegetation may change accordingly.

Another probable cause for the dieback and vegetation change could be increased salinity. Saltmarsh salinity, especially in the mid and the high marsh, is controlled by rainfall and evaporation rates (Ranwell, 1972). Since the rainfall over the study sites has decreased considerably over the past two decades while the temperature increased, salinity levels in the back marsh would have increased. Hence, the extensive bare areas recorded in the study, especially those that occurred in the high marsh or on ridges where they are well-drained, could be due to increased salinity rather than waterlogging. Bertness and Pennings (2000) support this postulation by suggesting that increasing evaporation with reduced rainfall can raise salinity levels to a tripping point over which plants can no longer persist and “climate-driven salt pans” will be formed. Salt pans can become hypersaline as a lack of shading by vegetation increases evaporation. This can potentially affect nearby plants and expand the salt pan. Salinity can also be affected by reduced groundwater flows (Thibodeau *et al.*, 1998).

Several studies have reported that large scale vegetation changes have resulted due to the effects of enhanced nutrients (eutrophication) in the saltmarsh (e.g. Bertness *et al.*, 2002; Pennings *et al.*, 2002). Bertness *et al.* (2002) found that increased levels of nitrogen facilitated the invasion of a low marsh dominant into the high marsh communities. On the terrestrial side, they found that enhanced nitrogen levels caused *Phragmites australis*, a cosmopolitan reed commonly found in brackish parts of many estuaries in Tasmania, to aggressively invade the high marsh. Another effect of nutrient loading is the proliferation of algal growth. Algal growth in saltmarshes is restricted by

nutrient availability, vegetative shading and temperature (van Raalte *et al.*, 1976). When there is an increased input of nutrients, unvegetated bare patches and warm temperature, algal growth can be excessive. Some study sites that were adjacent to irrigated farmlands had a significant amount of filamentous algal growth in the bare patches situated in the mid-high marsh. The algal matter was moved around by the tide and deposited on top of the saltmarsh vegetation, sometimes completely covering the plants. Over time, these areas might develop into “rotten spots” devoid of plant cover (Packham and Willis, 1997). Such rotten spots were a common feature within the study sites that were exposed to nutrient loading.

Internal marsh erosion can alter the substrate and topography of the marsh causing major vegetation changes (Allen and Pye, 1992). In many sites, especially in Duckhole Rivulet, the marsh pools were seen to be coalescing and forming extensive bare areas. The vegetation near these pools, especially where *T. arbuscula* occurred, appeared unhealthy. In the high marsh, many mud mounds were surrounded by bare patches with considerable algal growth. Similar patterns have been observed by Allen and Pye (1992) for marshes in Essex and Hartig *et al.* (2002) in New York City, with both studies attributing the marsh vegetation fragmentation and dissection to increased tidal inundation caused by sea level rise.

Cooper *et al.* (2001) have suggested that the sediment eroded from the saltmarsh edge and in the nearby estuarine/lagoonal area accumulates in the sub-tidal channel, and is then swept landwards by the flood tide and deposited in the inter-tidal zone (including the saltmarsh) within the inner estuary. Given that several patches of *S. australis* were recorded at the seaward edge and the mouths of tidal creeks, it can be hypothesised that some accretion has happened and that *S. australis* is an indicator of places where it had occurred. *S. australis* is known to be an effective coloniser of disturbed sandy substrates near the seaward/creek edge (Glasby, 1975) which are well-drained (Clarke and Hannon, 1971). The evidence for *S. australis* dominance indicating accretion was strongest in Pipe Clay Lagoon where, in some sections, the saltmarsh edges had visibly accreted with wind/wave deposited sand and gravel sheets. Most of this newly formed substrate was colonised by *S. australis* and subsequently taken over by *A. stipoides*. In the 1975 mapping, there were numerous occasions where *A. stipoides* occurred on ridges along the seaward edge. However, most of these patches have been lost to



erosion, while new ridges, largely colonised by *S. australis* formed where some of the material was deposited.

Pethick (1993) has suggested that the saltmarsh species composition in the inner estuaries will be altered with increased salinity caused by seawater incursion as a result of sea level rise. He noted that species characteristic of the brackish environment, including *Phragmites australis*, *Ficinia* spp. and *Juncus* spp., will be replaced by species adapted to more saline conditions while the brackish marsh species will move further up the estuary. Evidence for such a change occurring was recorded in Orielton Lagoon and Duckhole Rivulet. In Orielton Lagoon, the brackish marsh species that occupied the inner parts of Orielton Rivulet were replaced by more salt-adapted species, while the saltmarsh area increased in general in the upper estuary. In Duckhole Rivulet, *J. kraussii* rushland in the inner estuary was replaced by more salt-adapted communities. The change from *J. kraussii* rushland to more salt-adapted communities was also recorded widely across the study sites in the high marsh.

Saltmarsh transgression (or migration) in to the nearby low-lying farmlands constituted an “unplanned retreat” as opposed to “planned retreat” or “managed retreat” which is the human facilitation of the landward transgression of saltmarshes in the event of sea level rise (Hazelden and Boorman, 2001; Adam, 2002). It is expected that saltmarshes will retreat/migrate/transgress landward with an increase in sea level, with the extent of migration and hence the extent of saltmarsh determined by the tidal range. However, landward migration will be possible only where the land profile is gentle enough to allow access to tidal water. In places where uplands rise steeply enough from the marshes, or where sea walls exist, saltmarshes will not be able to move over the impediment and end up “squeezed” between the land and the sea. Natural barriers to landward migration occur in many of the saltmarshes of the study area (see Appendix II). In most such places, the grass/sedge/rush vegetation strip that bordered the marsh in 1975 became either *S. quinqueflora* or bare ground.

#### **4.3.1 Conclusion**

The saltmarsh vegetation has changed considerably over the study period with most identified changes reflecting the changes in environmental factors (Figure 4.26). Four main environmental factors possibly involved in these changes include: sea level rise;

climate change; nutrient inputs and land reclamation. Large areas of bare ground were recorded caused by the dieback of the saltmarsh vegetation possibly due to both the increase in tidal inundation and soil salinity. The low marsh vegetation has substantially increased in extent displacing the high marsh vegetation. A general shift was observed where vegetation lower in the tidal frame were moving up suggesting an increase in the tidal penetration caused by sea level rise. Increase in the tidal penetration could also have been responsible for the movement of saltmarsh vegetation up the estuary channel and transgression into low lying uplands. Changes were noted in the marsh edge vegetation possibly caused due to vertical accretion on the marsh surface.

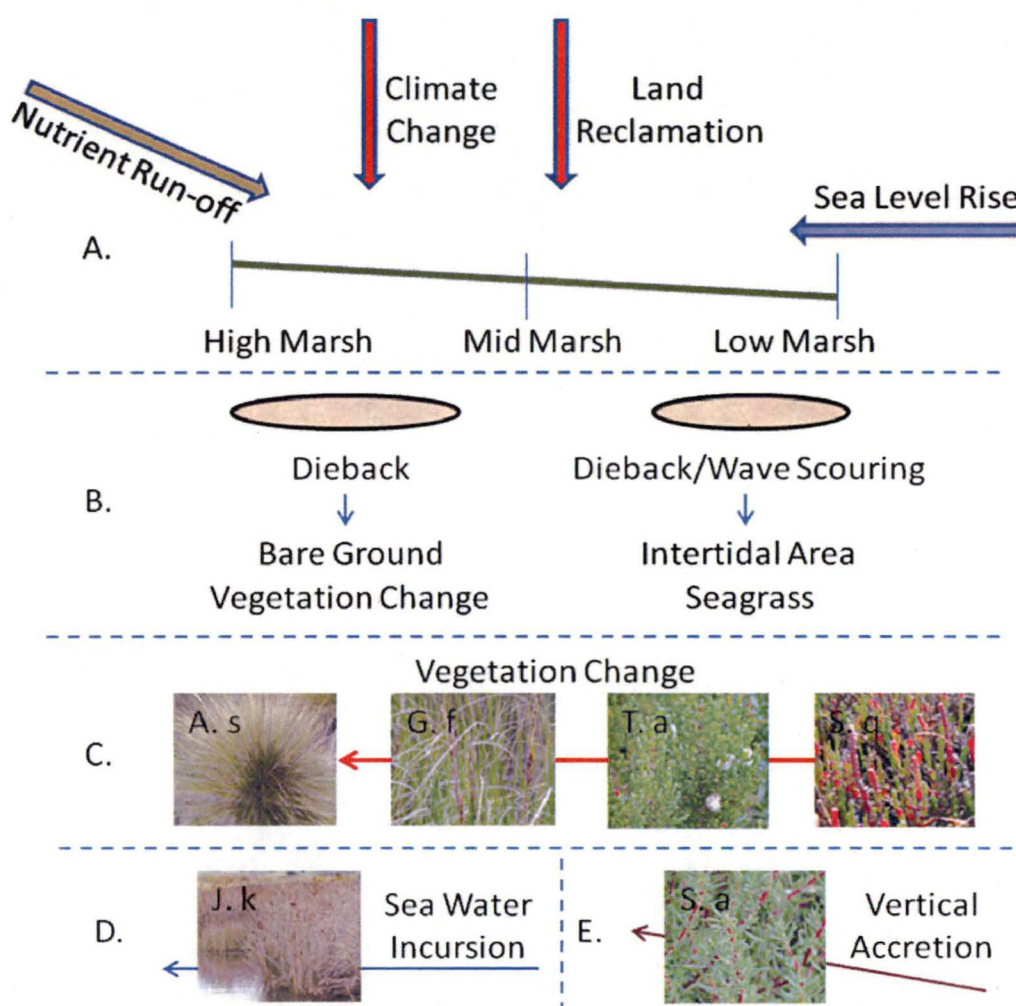


Figure 4.26. Major changes in the saltmarsh vegetation and forcing factors likely to be involved:  
A – Four main factors identified to affect vegetation change;  
B – Dieback in the low marsh giving way to seagrass and mudflats, while dieback in the mid-high marsh giving way to vegetation change and bare ground formation;  
C – The shift in vegetation dominance from low marsh species to high marsh marsh species;  
D – Sea water incursion as a result of increased tidal amplitude affecting *Juncus kraussii* dominance; and  
E – Possible vertical accretion assisting in the dominance of *Suaeda australis*.

## Chapter 5: Conservation and Management of Saltmarshes

### 5.1 Threats and Conservation Implications

There are many threats to coastal saltmarshes (Adam, 2002; Laegdsgaard *et al.*, 2009). Some of these threats, such as sea level rise, are more prevalent than others, such as mangrove incursion, and vary in their incidence between bioregions and, in some cases, even between individual saltmarshes. The wide range of threats facing the saltmarshes in south east Tasmania can be grouped into two major groups: anthropogenic pressures; and environmental change.

#### 5.1.1 Anthropogenic Pressures

The biggest anthropogenic threat to the saltmarshes in the study area so far has been land reclamation. Saltmarshes have been viewed as wastelands that provide no economic return and have been reclaimed for a various purposes, including for agriculture, buildings, roads and golf courses among others. Over 300,000 m<sup>2</sup> of saltmarsh area has been lost to land reclamation alone within the Pitt Water region in just over 40 years. Within the study sites, the loss amounted to nearly six per cent of the 1966 saltmarsh area. Land reclamation is mainly perpetrated by dumping or by tidal restriction or a combination of both. Given this trend of land reclamation in the region, with increasing sea levels, many landowners who have their property boundary at high water mark can be expected to further raise tidal barriers to safeguard their lowlands. This may effectively leave several saltmarshes in the region “up against the wall” as sea levels rise.

The second major anthropogenic threat facing the saltmarshes is eutrophication, a stressor hitherto largely ignored in the literature on Tasmanian saltmarshes. Large areas of saltmarshes within Pitt Water area are affected by a significant amount of filamentous algal growth caused by the addition of nutrients from either surface run-off from nearby agricultural lands or groundwater flows. Saltmarshes are known for their important function of nutrient capture from surface and ground waters before they reach the seagrass meadows and estuaries/lagoons (Valiela *et al.*, 2000; Valiela and Cole, 2002). When excessive amount of nutrients are released into the marsh, the regular

nutrient sequestration function of the marsh is compromised thereby also increasing nutrient loads in the receiving waterways. Eutrophication has been suggested to be the biggest reason for the widespread loss of seagrasses around the world (Burkholder *et al.*, 2007). Within Pitt Water, algal growth caused by nutrient inputs has caused a decline in seagrasses by about 94%, in the period between 1950 and 1990 (Rees, 1994).

Grazing is another major issue for saltmarshes in south east Tasmania. Some sites have been grazed by sheep, cattle and horses. Grazing by hard-hoofed animals can considerably change the soil characteristics and have a detrimental effect on saltmarshes. Grazing by rabbits is also common in the Pitt Water saltmarshes, with several rabbits and rabbit burrows noted during field surveys. Rabbit grazing has been noted to be a threat to saltmarsh revegetation (Parks and Wildlife Service, 2009). Gouldthorpe (2000), while documenting the negative effects of grazing on some Pitt Water and South Arm saltmarshes, noted that the problem might diminish in the future as landowners were undertaking measures to reduce grazing impact. However, any measures taken have been few and far between as grazing still continues in many saltmarshes.

Dams and reservoirs are a considerable threat to coastal saltmarshes. They reduce the flow of freshwater and sediment inputs into the marsh and affect the regular ecosystem processes. Almost every major creek draining into the saltmarshes in the Pitt Water area have been dammed, some in more than one place. Many creeks have been dammed very close to the creek mouths, in the process, eliminating the saltmarshes at the creek mouth, and also removing the possibility of the marshes moving up the creek as sea level rises. Groundwater availability is another threat to saltmarshes, as reduced groundwater flow can increase soil salinity (Thibodeau *et al.*, 1998) and excessive groundwater extraction can cause saltmarsh subsidence (Kennish, 2001). While there is a large number of groundwater extraction points in Pitt Water and South Arm area, the quantity and effect of extraction is not known.

A major issue for some saltmarshes in northern Tasmania is the effect of introduced/invasive species such as *Spartina anglica* (Rice Grass; Kriwoken and Hedge, 2000). However, the grass was not recorded within the study sites and is possibly not a major threat in the near future. Other common weeds that commonly occur on the



terrestrial boundary of the marsh include *Plantago coronopus* (Buckshorn Plantain) and *Lycium ferocissimum* (African Boxthorn), but are uncommon or non-existent within the marsh. Some exotic species occur on disturbed patches within the marsh, such as *Senecio elegans*, but are not common within the Pitt Water and South Arm areas. Therefore, weeds are not a major problem for the saltmarshes in the study sites.

While all the threats discussed above are widely applicable for saltmarshes in non- or sparse-residential areas, the major threats within residential areas are different and mainly include stormwater drains, littering and recreation. Of these, stormwater drains constitute one of the most important threats to saltmarshes. Drains bring in freshwater often rich in nutrients. This can change the community composition of the saltmarsh and facilitate the creation of “rotten spots” within the marsh with immoderate algal growth. Another common feature of the saltmarshes near residential areas is the amount of litter found in the marsh, ranging from bottles and cans to old cars. Littering can smother saltmarsh plants and facilitate the invasion of weeds into the marsh (Laegdsgaard *et al.*, 2009).

Recreation on the saltmarsh can take many forms and in each form affects the marsh in some way. The worst damage is caused by four-wheel drive and other off-road vehicles which squash vegetation and leave depressions within the marsh changing the drainage characteristics. Continued use causes soil compaction of extensive areas making it uninhabitable for saltmarsh plants and fauna (Kellaway, 2007). Use of cycles also causes similar effects but is much more muted. Horse riding is a popular recreation in Pipe Clay Lagoon and horse-hoofs dig up the saltmarsh substrate and disturb the vegetation and soil. Continuous trampling caused by walking can cause soil compaction and create bare spots (walking tracks) within the saltmarsh.

### **5.1.2 Environmental Change**

Climate change and sea level rise constitute the biggest long term and widespread threats to saltmarshes. Evidence obtained from this study on the change in extent and vegetation composition of the saltmarshes in south east Tasmania show that marshes have suffered considerable lateral erosion, internal dissection, vegetation dieback, ponding and soil salinisation. Lateral erosion has caused a net loss of saltmarsh area of nearly 130,000 m<sup>2</sup> (which is nearly four per cent of the 1996 saltmarsh area) while the

other abovementioned processes have resulted in a significant increase in bare areas and large scale changes in vegetation dominance patterns. Bare areas signify a loss in biomass and decreased saltmarsh function and productivity. Large scale vegetation changes can alter the habitat available for fauna and reduce the biodiversity and ecological function of the saltmarsh (Hughes, 2004; Laegdsgaard, 2006).

The annual average temperature in most parts of Australia is predicted to increase by 0.4–2.0 °C by 2030 and 1.0–6.0 °C by 2070 with the greatest warming occurring in spring (Australian Commonwealth Scientific and Research Organization, 2001). The CSIRO models indicate that rainfall for Tasmania will decrease in spring, summer and autumn (–10% to +5% by 2030 and –35% to +10% by 2070) while winter will have increased rainfall (–5% to +20% by 2030 and –10% to +60% by 2070). Evaporation is expected to increase with higher temperatures, with up to a 12% increase per degree of global warming predicted for Tasmania (Australian Commonwealth Scientific and Research Organization, 2001).

The predicted rises in temperature and warmer summers in Tasmania will have a detrimental impact on saltmarshes (Hughes, 2003). This impact will be further exacerbated by reduced rainfall. The main reaction in the saltmarsh to these impacts will be an increase in salinity that could have severe repercussions for the saltmarsh vegetation (Bertness and Pennings, 2000). The vegetation patterns in south east Tasmanian saltmarshes are directly linked to salinity and drainage (Kirkpatrick and Glasby, 1981). Changes in salinity will alter the vegetation composition of the saltmarshes and increase bare areas as was evidenced from the present study. Climate change may also alter the geographical distribution of species and certain frost sensitive species that are restricted to lower latitudes can extend their distribution towards higher latitudes when favourable conditions emerge due to climatic warming (Adam, 2002).

Global warming will increase the problems associated with excess nutrients in the saltmarsh, as increased temperature with reduced rainfall will proliferate the productivity of “nuisance algae” (Welch *et al.*, 2001). Filamentous algae caused vegetation dieback and bare ground creation may become intensified and can potentially destroy large areas of saltmarshes with the projected climate change predictions for Tasmania. The effect of eutrophication, temperature rise and increased CO<sub>2</sub> in the

atmosphere may affect C<sub>3</sub> and C<sub>4</sub> (photosynthetic pathways) species differently and favour particular species over others which fail to adapt to the changed environment (Gray and Mogg, 2001; Adam, 2002). However, as each saltmarsh plant varies in its adaptation and competitive abilities, any predictions of their response to climatic changes and nutrient loading is hard to make without experimental evidence.

Sea level rise has caused large scale vegetation and physiographical changes within the saltmarshes. As predicted with increased waterlogging, the low marsh species have replaced the high marsh species as the dominant vegetation community. Large areas of bare and often waterlogged spots caused by internal erosion and vegetation dieback were recorded. With an increase in sea level rise, further losses of high marsh species and an expansion of bare area can be expected (Allen and Pye, 1992; Donnelly and Bertness, 2002). Pitt Water area is known to be one of the most important areas in Tasmania for the mid-high marsh halophyte *Tecticornia arbuscula* (Kirkpatrick and Glasby, 1981). Sea level rise will pose considerable threats to the large Pitt Water populations of *T. arbuscula* in the mid marsh while global warming will be a threat in the high marsh, mainly by increasing salinity. In the long term, populations of *T. arbuscula* can be expected to dwindle if landward migration does not happen. Pitt Water sites have a high proportion of the Tasmanian populations of two rare saltmarsh species *Limonium australe* and *Wilsonia humilis*, which can be highly endangered if saltmarsh losses continue.

A common phenomenon associated with sea level rise is the loss of the tall and long lived high marsh plants which are replaced by short low marsh plants (Huiskes, 1990). The loss of high marsh plants, and the general loss of saltmarshes, has several impacts for migratory and shore birds in south east Tasmania. The saltmarshes and the adjacent mudflats in the Pitt Water and South Arm areas are one of the most important habitats for several migratory birds and resident shorebirds in Tasmania, some of which are endangered (Bryant, 2002; Birds Tasmania, 2006; Parks and Wildlife Service, 2009). Birds use the tall high marsh plants to avoid strong winds and predators and roost in the mid-high marsh at high tide when the mudflats are tidally submerged (pers. obs.; Spencer *et al.*, 2009). The loss of the high marsh plants will reduce the habitat quality available for birds. The south east Tasmanian saltmarshes are also used by waterfowl and some terrestrial birds such as *Neophema chrysostoma* (Blue-winged Parrot) which

feed on *Sarcocornia quinqueflora* seeds (Park, pers. comm.). Large scale losses of saltmarshes, as predicted in the inundation models, will affect the populations of these birds.

The composition of the crustaceans and molluscs of the Tasmanian saltmarshes depend on the “geomorphological, topographic and tidal conditions which control the degree of emergence and submergence of a marsh” (Richardson *et al.*, 1998). A change in the tidal condition and internal marsh erosion will negatively affect the species composition of the detritivores and reduce their role in the marsh function.

Saltmarshes have an important role in the coastal landscape by exporting rich organic matter into subtidal food webs and supporting commercially and recreationally important fish and crustaceans (Valiela *et al.*, 2000; Deegan *et al.*, 2000). Both Pitt Water estuary and Pipe Clay Lagoon is popular for recreational fishery and commercially important for marine farming (DPIWE, 2001; DPIW, 2008). Any loss of saltmarshes in these areas may affect the fisheries and marine farming. Saltmarshes also help regulate the nutrient (both in surface and ground water; Harvey and Odum, 1990) and sediment input into the estuarine areas. Pitt Water estuary has been enriched with nutrients and sediments from nearby agricultural land. This has resulted in the reduction of clam, oyster beds, ostracods and calcareous foraminifera (Lewis, 2006). Further losses of saltmarshes will increase nutrient levels and sedimentation in the estuary and may lead to further losses in the intertidal and benthic fauna.



## **5.2 Policy and Management Implications**

Recognising the role that saltmarshes play in human wellbeing and regional economics (Mazumder, 2004), urgent action is required in terms of policy and management to reduce the pressures acting on saltmarshes and to maintain their area along the Tasmanian coast, especially within important estuaries and lagoons. Several planning and management initiatives have been suggested for Australia (Harty, 2004; Laegdsgaard *et al.*, 2009) and elsewhere (Doody, 2008). Some initiatives that could be applicable to south east Tasmanian saltmarshes are discussed in this section.

### **5.2.1 Catchment Management**

Saltmarshes lie at the receiving end of the whole catchment. Any major environmental changes in the catchment will have an effect on saltmarshes and in some cases, as seen in Pitt Water, the effect can be pronounced. Hence any catchment management strategy should consider the impact of human action in the catchment with respect to coastal saltmarshes. The main catchment scale impacts on the saltmarshes are related to nutrients, sediments and freshwater. The control of nutrient inputs into the saltmarsh is an important catchment scale management issue and needs to be controlled by both land use planning and agricultural practices. Protected Environmental Values have been set for the Pitt Water and South Arm regions to maintain water quantity and quality in the region and support fisheries, marine farming and recreation (DPIWE, 2003b). Large scale loss of saltmarshes will directly affect these activities. Hence, management actions aimed at protecting environmental values of these coastal waterways (e.g. Natural Resource Management, 2005) need to consider saltmarshes both as a management solution and an indicator. Saltmarshes can provide as a useful indicator of natural resource management effectiveness in the catchment and are relatively easy to monitor especially in the case of nutrients.

### **5.2.2 Habitat Restoration**

A recent increase in the profile of saltmarshes in Australia has been followed by several saltmarsh restoration initiatives aimed at restoring and creating saltmarsh habitats (Streever, 1997; Laegdsgaard, 2006). One of the most notable projects was the Kooragang Wetland Rehabilitation Project carried out in the Hunter Estuary, New South Wales. The project involved the restoration of saltmarshes by removing tidal barriers, creation of new areas of saltmarsh by excavating degraded pastures and controlling/excluding grazing (Streever, 1998). Large areas within Pitt Water estuary

that are under private freehold are tidally restricted and grazed. The restoration of these areas will involve providing support for the landowners either in terms of fencing infrastructure or monetary compensation for providing for the public good. Restoration in public land needs effective land management that includes the exclusion of recreational vehicles and hard-hoofed animals and providing designated pathways for walking within the saltmarsh. Restoration of the eroding marsh can be done by promoting accretion or reducing wave energy at the saltmarsh periphery by offshore breakwaters and rip-rap (Doody, 2009). However, the long term effectiveness and the economics involved in these operations are not fully understood and will vary considerably between different coastal environments.

### **5.2.3 Planned Retreat**

Planned or managed retreat has emerged as an important policy response to saltmarsh loss caused by sea level rise (Boorman, 1999; Adam, 2002; Townsend and Pethick, 2002). It mainly involves non-interference with the transgression of saltmarsh plants inland as sea level rises. In some cases, this process is facilitated by removing existing tidal barriers or excavating land to allow the landward retreat of saltmarshes. Hazelden and Boorman (2001) have reported that saltmarshes are able to successfully germinate following tidal inundation and sedimentation of low lying agricultural uplands. Landward retreat was recorded within some sites in Pitt Water area, where saltmarsh plants transgressed into low lying agricultural land as a result of increased tidal penetration. With an increase in sea level, the volume of tidal water entering the estuary and lagoon will increase thereby increasing the high tide mark and the area of the estuary. Hence Pethick (1993) suggests that “[m]anagement strategies should be developed to allow intertidal profiles to migrate landwards and to increase the width of estuary channels.” Planning for landward retreat involves several considerations:

1. Local topography and tidal access to uplands (retreat pathways): These can be identified by inundation modelling, which can predict where the tide mark will be in the future with a predicted sea level rise, assuming no vertical accretion. Saltmarsh plants follow the tide and colonise tidally inundated areas as glycophytes die with saltwater incursion (Choi *et al.*, 2001).
2. Land value and land tenure: The value of the upland should be relatively low, such as grazing pastures, for the retreat to be cost effective. In some cases the

upland may fall within Crown Land or reserves (such as Orielton Lagoon) and can be used for saltmarsh retreat without cost.

3. Wave exposure: Saltmarshes in retreat areas are more likely to survive where wave exposure is low, as in the inner estuaries (Pethick, 1993).
4. Longshore drift and other costal sedimentation processes: As seen from the present study, some parts of the saltmarsh shoreline may accrete vertically and expand laterally. Retreats adjacent to such accreting shorelines will be more effective.
5. Retreating up the creek/river: Pethick (1993) has suggested that upper estuaries will be the refugia for saltmarsh species in the event of accelerated sea level rise.

#### **5.2.4 Carbon Sequestration**

Tidal wetland soils have high carbon sequestration rates and can be considered as valuable carbon sinks (Rabenhorst, 1995). Any carbon sequestration program could benefit by considering investing in saltmarshes as they are known to accumulate more carbon per unit area in their soils than peatlands, forests and agricultural soils (Choi *et al.*, 2001; Chmura *et al.*, 2003; Hussein, 2004). Hussein (2004, p. 1794) reported in their study that “C sequestration in mineral soils of agro and upland forest ecosystems is generally of limited capacity and tends to reach steady-state condition within relatively short time” while in saltmarshes “C sequestration will continue to occur with time by accumulation in the organic horizons, and with increasing storage capacity.” Furthermore, Choi *et al.* (2001) and Andrews *et al.* (2006) have suggested that facilitating the landward migration of the saltmarshes in the event of sea level rise will increase their carbon storage potential and be economically beneficial, especially in the long term.

#### **5.2.5 Conservation Protection**

Saltmarsh conservation by inclusion into the protected area estate (both public and private) may be an effective way to enhance, manage and sustainably use saltmarsh resources (Laegdsgaard *et al.*, 2009). In the Pitt Water region, Pitt Water Nature Reserve covers only sites 6, 13 and parts of 8, 10 and 15, while the majority of the saltmarsh areas lie outside the reserve. Of the saltmarshes that lie outside the reserve, site 16 occurs on unallocated Crown Land which could be incorporated into the reserve. Site 2 and parts of Site 12 are within private reserves while all other sites lie within private agricultural land. Saltmarshes within private land, especially those that were

identified in the inundation modelling to have the potential to move upland in case of a 1.1 m sea level rise at the turn of the century, could be identified as key investments for saltmarsh conservation (and carbon sequestration) into the future. The current Pitt Water Orielton Lagoon Ramsar Site boundary does not include large areas of saltmarsh. Conservation planning of the Pitt Water region, which is nationally considered as a high conservation value aquatic ecosystem, needs to focus on the linkages and buffer/management zones that lie outside the core areas (Auricht *et al.*, 2009). In this respect, saltmarshes form an important buffer zone for protecting the core values of the estuary and needs to be incorporated within the management of the estuary as a whole. Hence, it is recommended that saltmarshes be included within the Ramsar Site and be afforded legislative and management protection.

### 5.2.6 Legislation

The existing Australian State (including Tasmania) and Territory legislation and policies pertaining to saltmarshes have been summarised by Laegdsgaard *et al.* (2009). The three Tasmanian state legislation directly relating to saltmarshes are the *Nature Conservation Act 2002*, *Forest Practices Act 1985* and *Threatened Species Protection Act 1995*. Other legislation affects saltmarshes indirectly through land use planning and approval, environmental impact assessment, pollution control and water quality management. The *Nature Conservation Act 2002* lists “saline aquatic herbland” (AHS – TASVEG Code) and “wetland (undifferentiated)” (AWU) as a Threatened Native Vegetation Communities (TNVC) (DPIW, 2009). Both these community types primarily include seagrasses and brackish marsh species and do not apply to all of the dominance communities present in Pitt Water and South Arm saltmarshes. The four main vegetation types that directly apply to saltmarshes are “succulent saline herbland” (ASS), “saline sedgeland/rushland” (ARS), “saltmarsh (undifferentiated)” (AUS) and “coastal grass and herbfield” (GHC) (Harris and Kitchener, 2005). However, these communities are not included as TNVC. Considering the loss of *T. arbuscula* shrubland (which falls under ASS and AUS) and the impending threats of sea level rise, climate change and anthropogenic pressures, the ASS community type need to be considered as TNVC and afforded subsequent conservation measures associated with the listing. *Gahnia filum* and *Juncus kraussi* fall under ARS and *Austrostipa stipoides* and *Poa labillardieri* fall under GHC, both not under severe pressure as compared to ASS.



### **5.2.7 Communication, Education and Public Awareness**

The effectiveness of policy and management initiatives for saltmarsh conservation may require understanding and support from the community. For this reason, communication, education and public awareness (CEPA) of wetland values has been widely recognised as an important tool in sustainably managing wetlands (Rose and Bridgewater, 2003; Ramsar Convention Secretariat, 2006). CEPA initiatives include setting up saltmarsh education and interpretation centres, disseminating information about the values of saltmarsh ecosystems and enabling local action for promoting the sustainable use of saltmarshes. While government and non-governmental organisations have been involved in promoting the education and awareness of saltmarsh values in South Arm and Pitt Water areas, a specialised saltmarsh education and interpretation centre would complement other CEPA efforts. Ralphs Bay, with its diverse saltmarsh vegetation communities, extensive mudflats and large year-round waterbird populations could provide as an excellent location for an education centre. Orielton Lagoon, with its extensive saltmarshes and birdlife, could also be used as a saltmarsh education centre.

### **5.2.8 Monitoring and Enhancing Knowledge**

South east Tasmanian saltmarshes have changed considerably over the past three decades and with increasing pressures, the changes can be expected to continue into the future. Hence, it is important to monitor the changes and inform policy and management in a timely basis. Monitoring the vegetation condition and response to climate change, sea level rise, catchment management and on-site management need to be carried out regularly to identify short-term vegetation responses to environmental factors. The effect of nutrients, both from surface water and groundwater, on the saltmarsh vegetation and community composition is a critical aspect that needs further research. Given that saltmarshes have eroded rapidly, the rates of erosion and sedimentation need to be monitored to identify future trends in shoreline retreat/advancement. Monitoring process should select both areas of high wave exposure where severe erosion has been recorded and areas where shoreline advancement was recorded. Since Tasmanian saltmarshes generally fall within two substrate types, sand/gravel or clay/silt, both substrate types should be monitored to better understand the relationship between substrate type and erosion. While erosion was the dominant process in saltmarshes, some areas of accretion were recorded. While wave exposure was one major factor in determining erosion and accretion, other factors may be involved that needs to be

studied to understand the sediment dynamics of the shallow coastal environments in a time of change.

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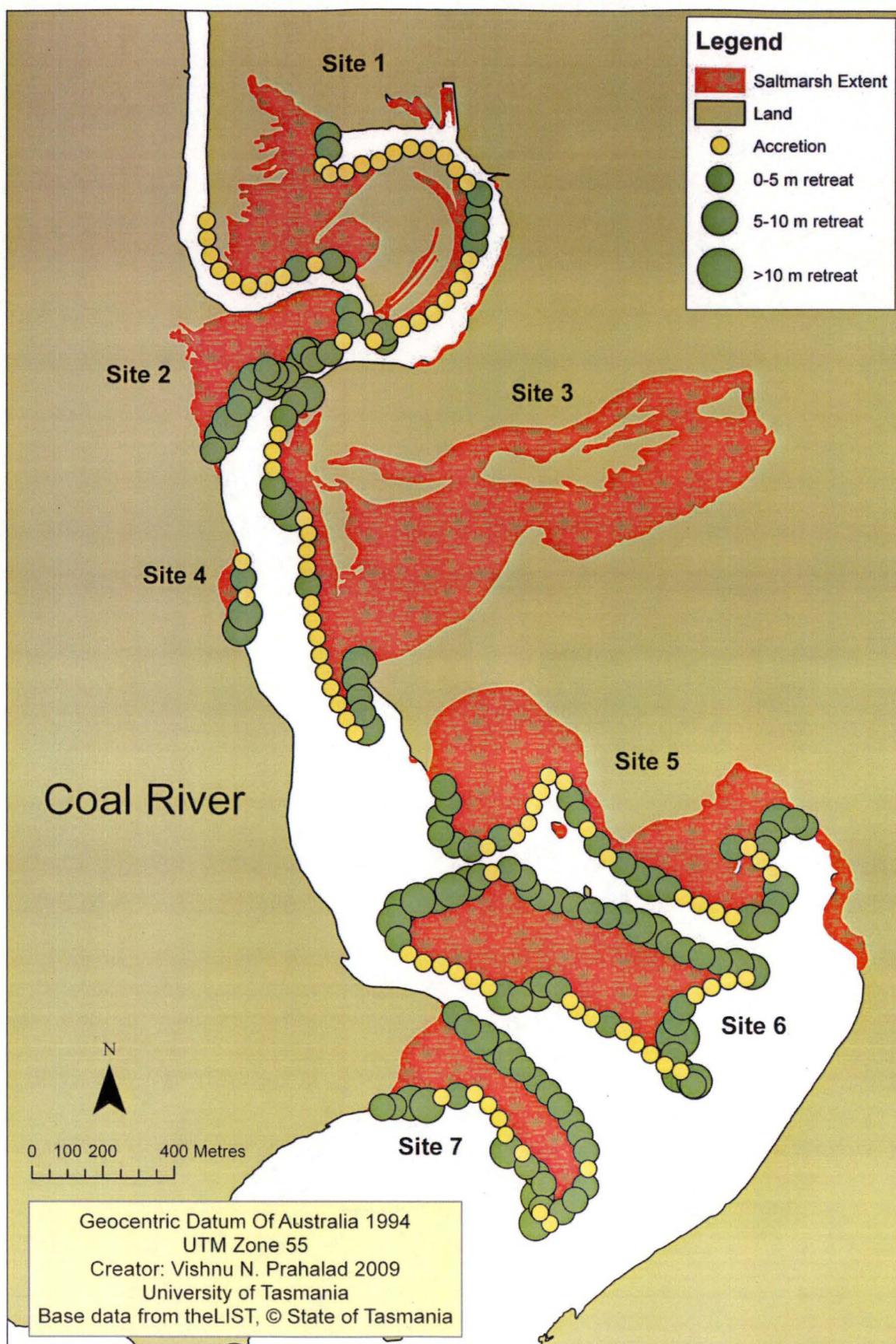
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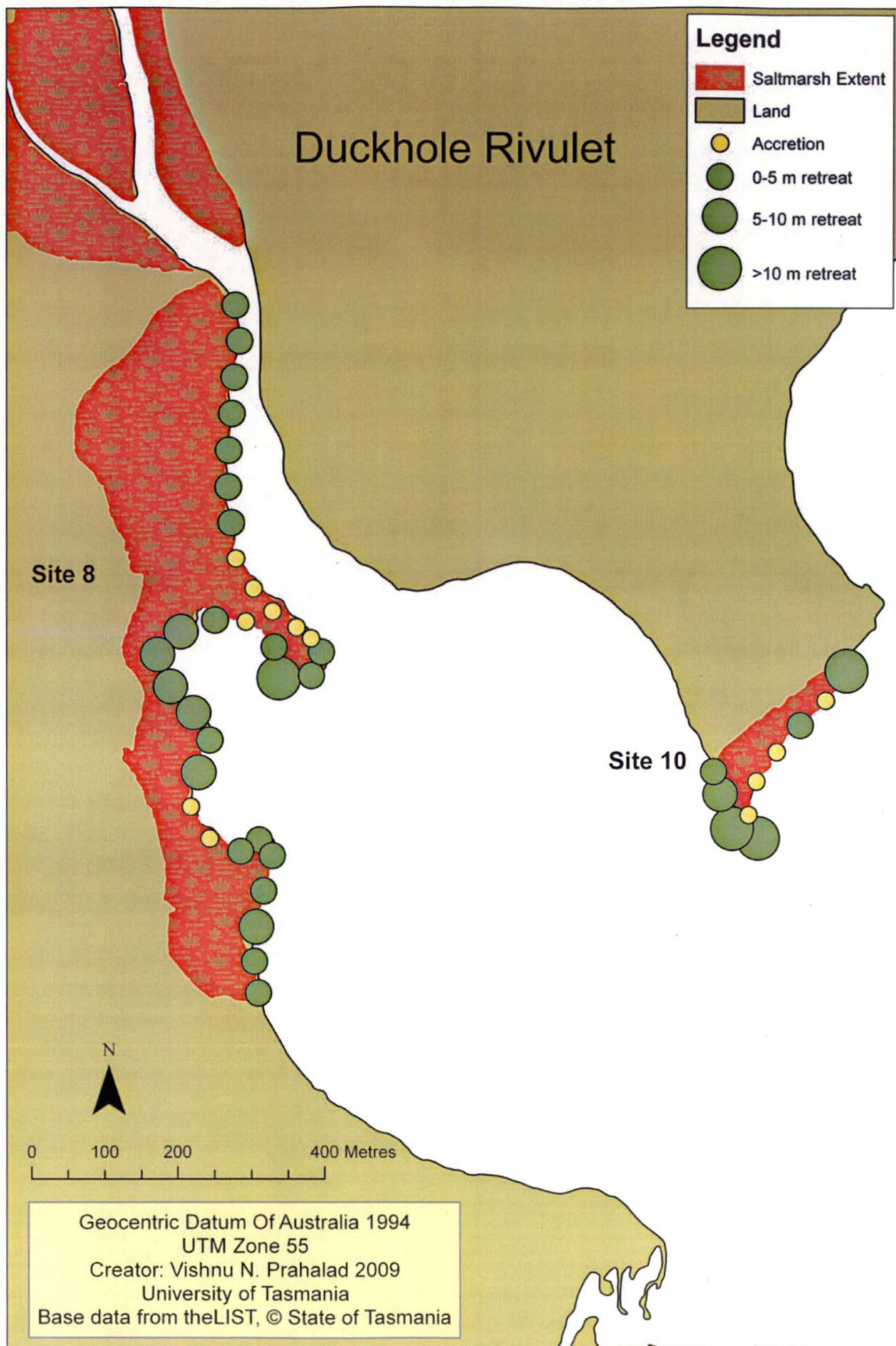
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## **Appendix I: Saltmarsh Shoreline Advancement/Retreat**

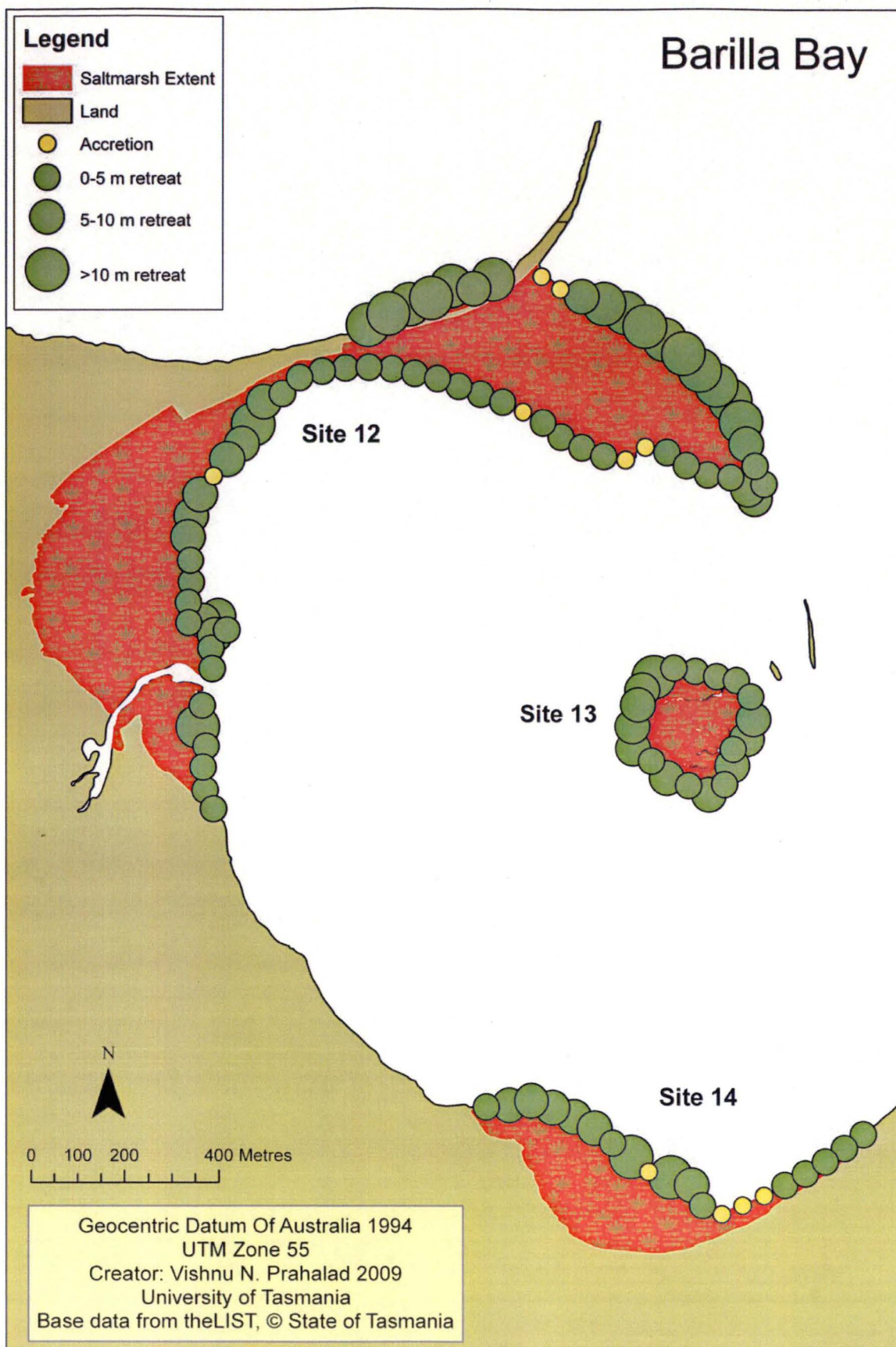
Saltmarsh shoreline advancement/retreat maps provided here indicate the parts of the shoreline (at each sampling point) that had eroded and parts that had accreted. A total of 607 sampling points, digitised on the 1975 coastline at 50 m intervals, were used. Four groups have been employed to indicate accretion and degree of erosion. They include: advancement; 0-5 m retreat; 5-10 m retreat; and >10 m retreat. Map for Orielton Lagoon has not been included as the shoreline retreated uniformly in all places possibly as a result of the changes in tidal regime after the opening of the Lagoon for tidal flushing.

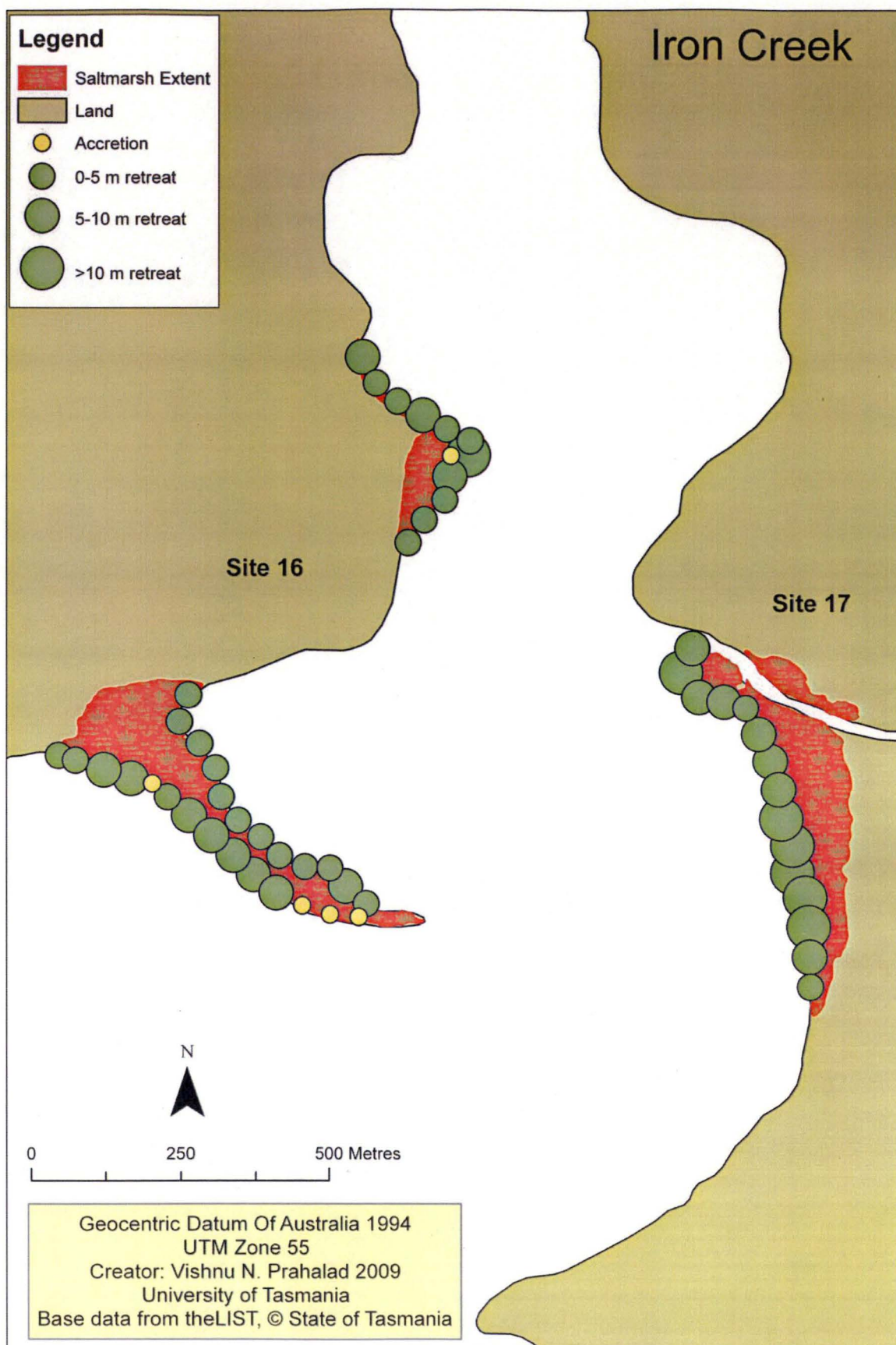




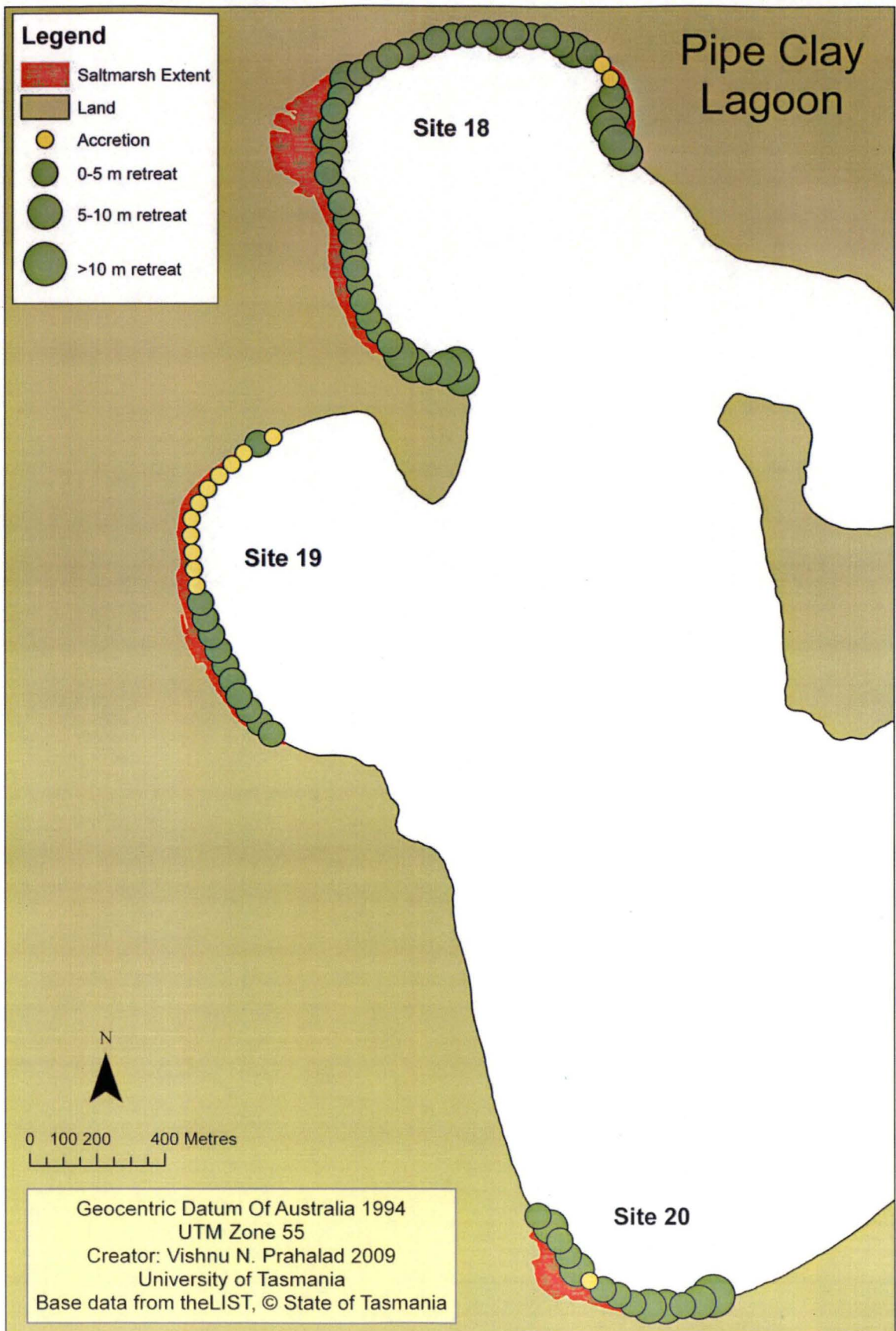


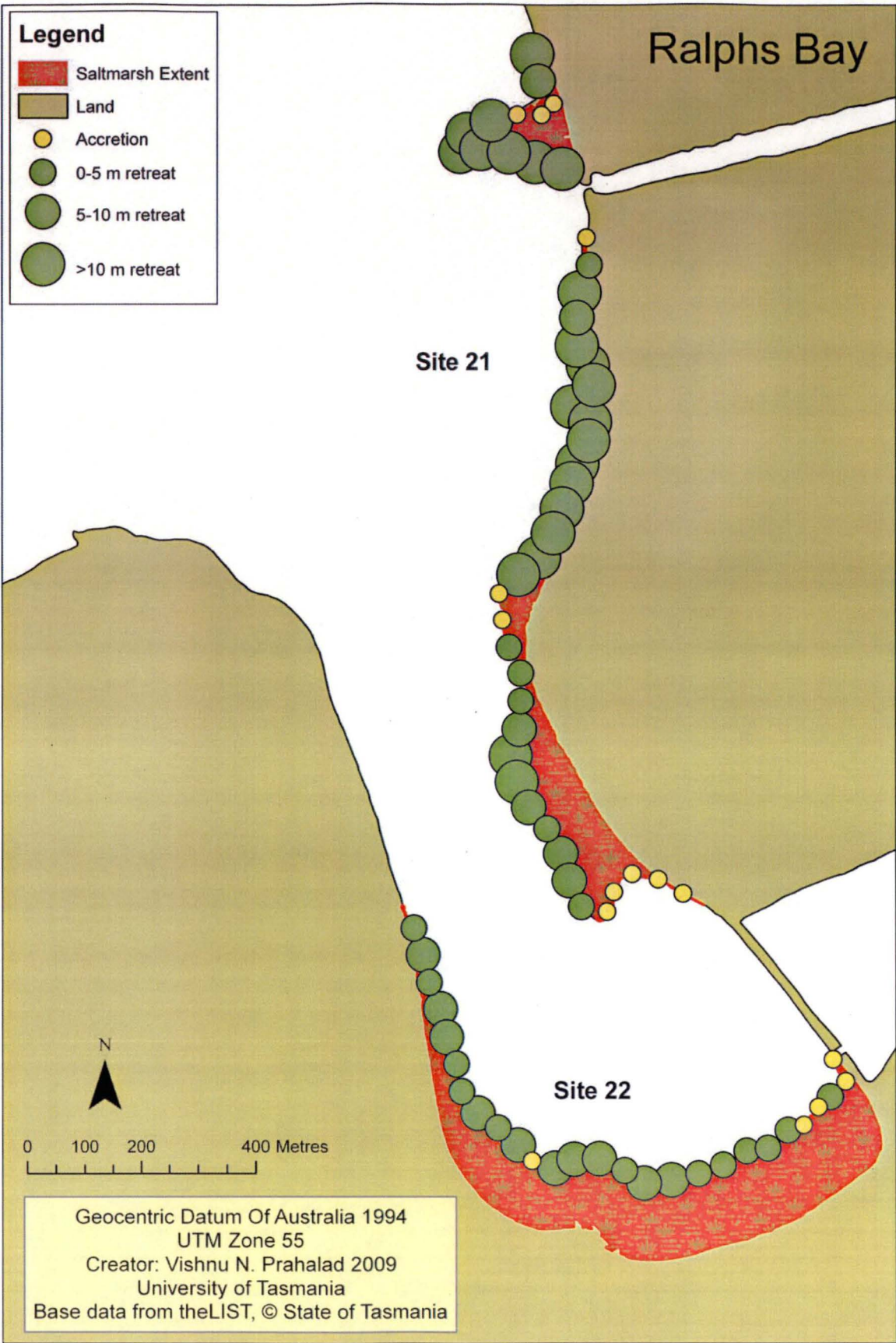








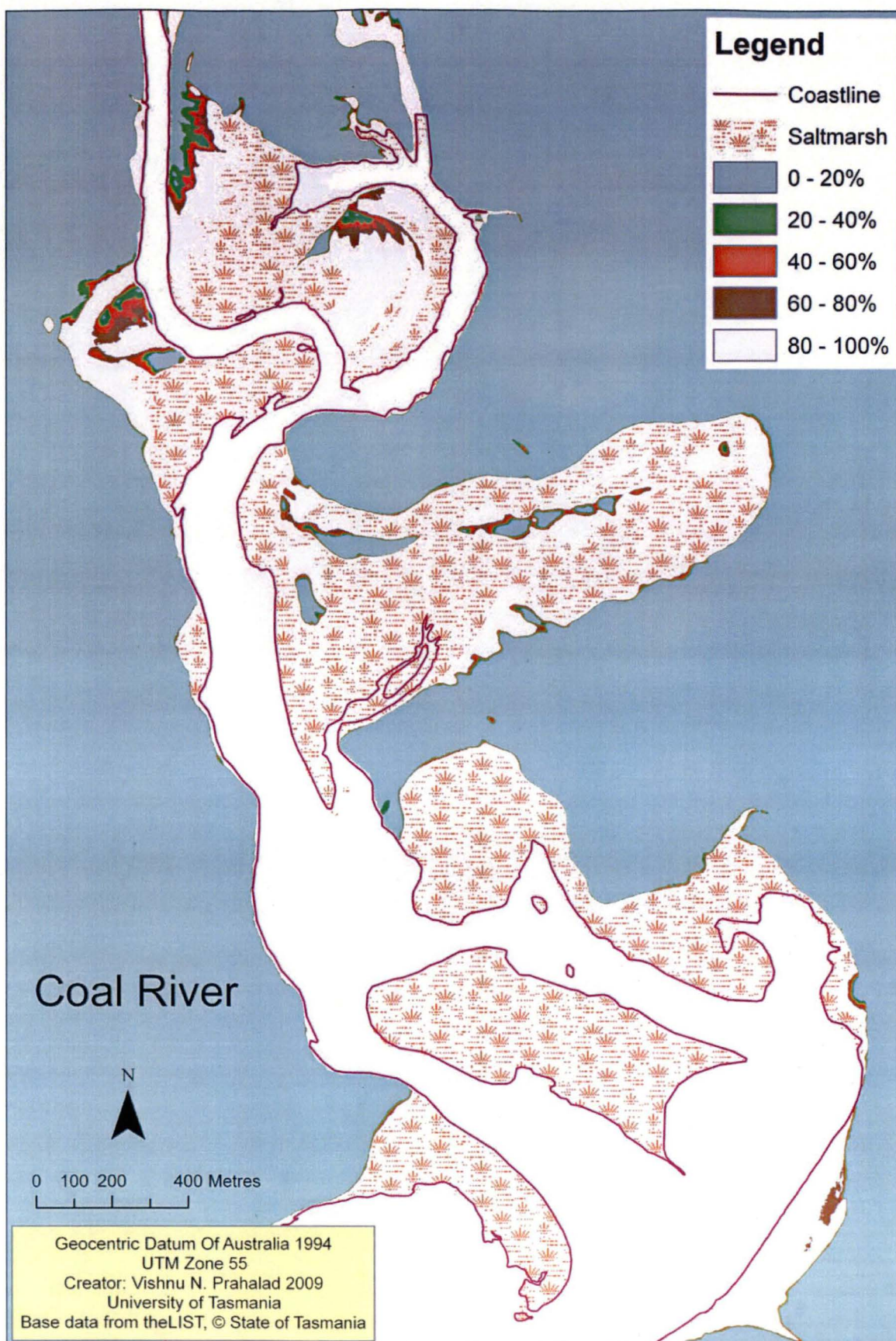




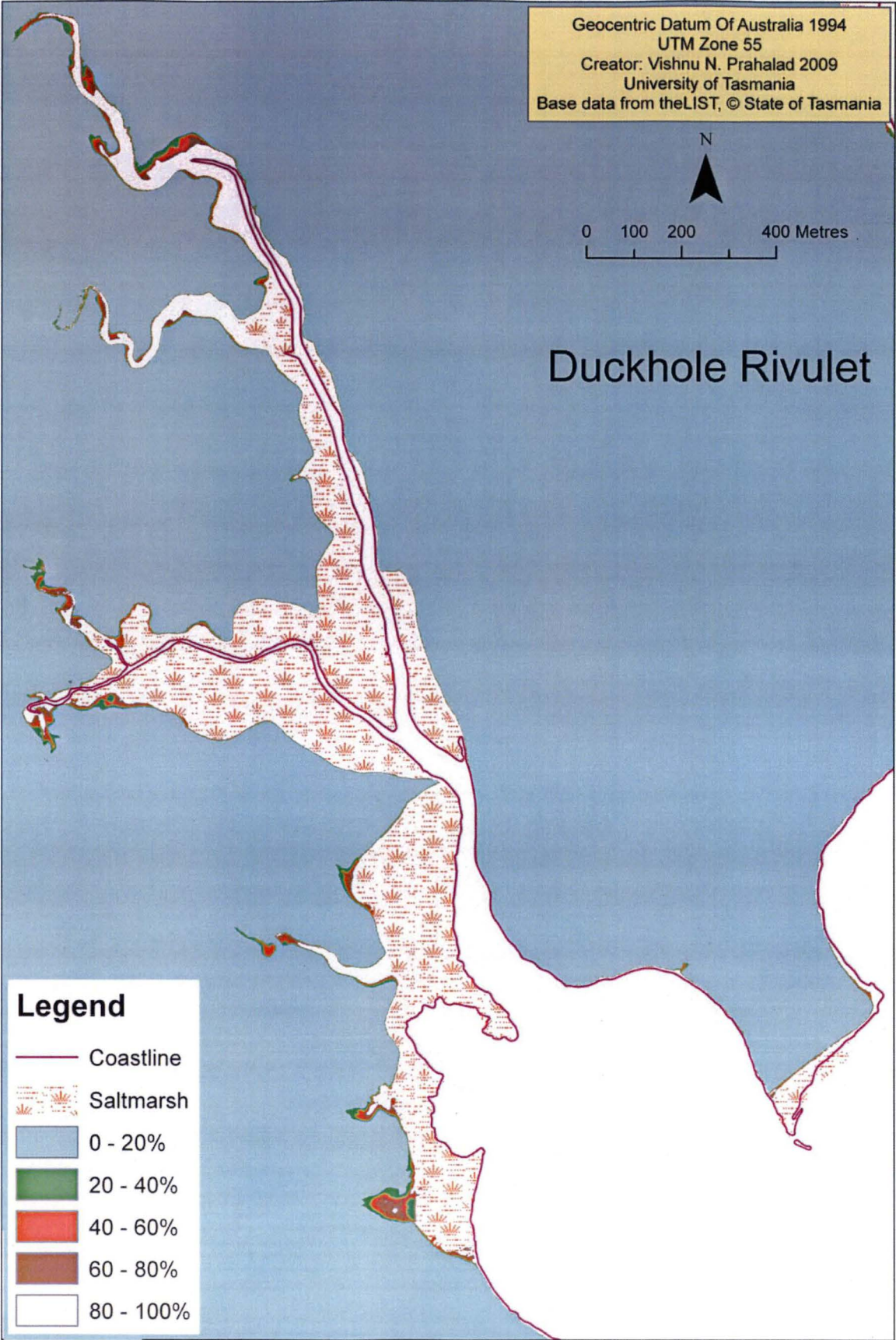


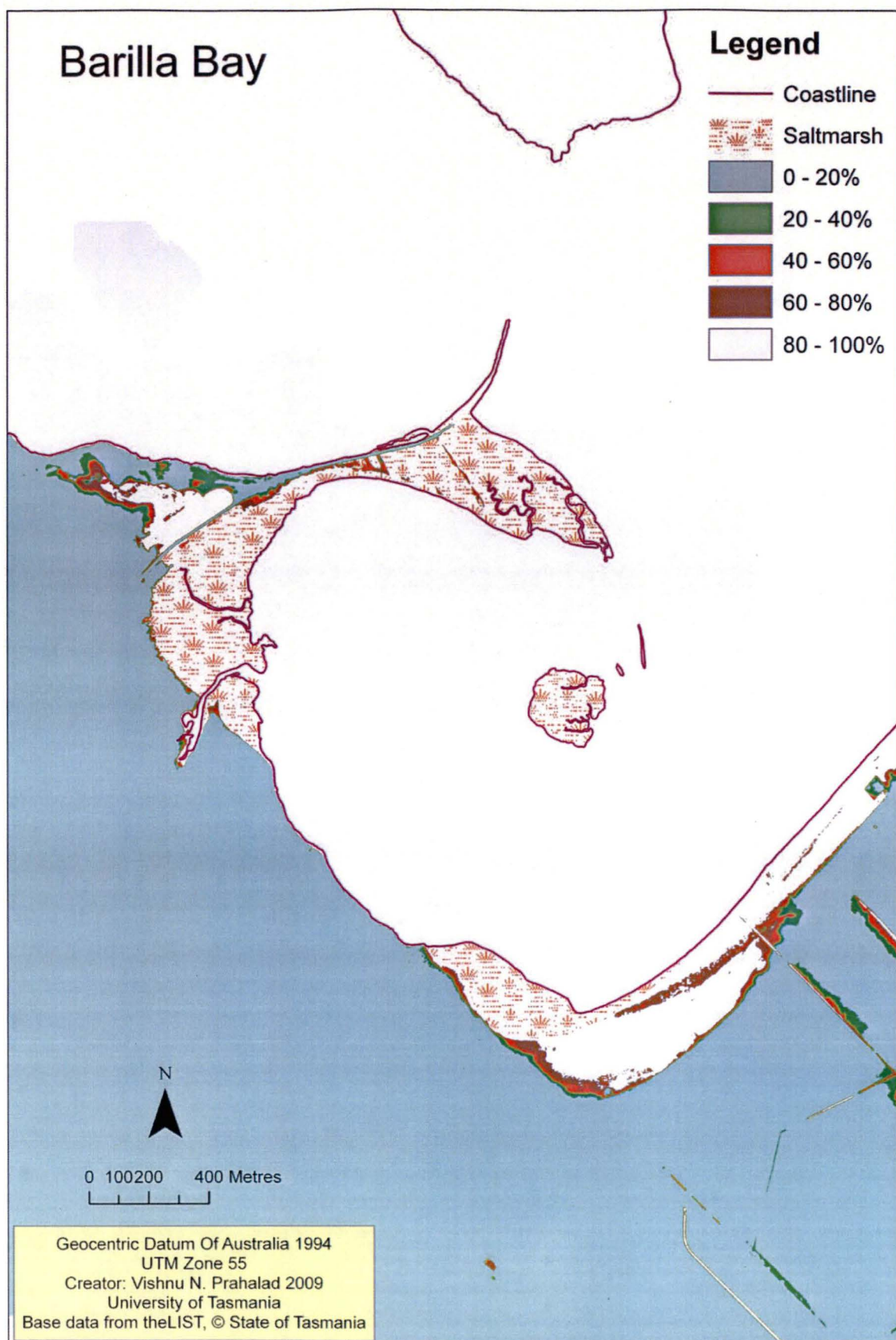
## **Appendix II: Inundation Modelling Maps**

Maps from the inundation modelling have been presented here. Maps indicate the predicted areas of land inundated by the flood tide at mean high water mark in 2100 given a 1.1 m sea level rise scenario for each of the seven study areas. There are five different probabilities shown: 0-20%; 20-40%; 40-60%; 60-80%; 80-100%. The probabilities are derived from the known errors in the input datasets, which includes data capture and processing errors in the LiDAR digital elevation model.





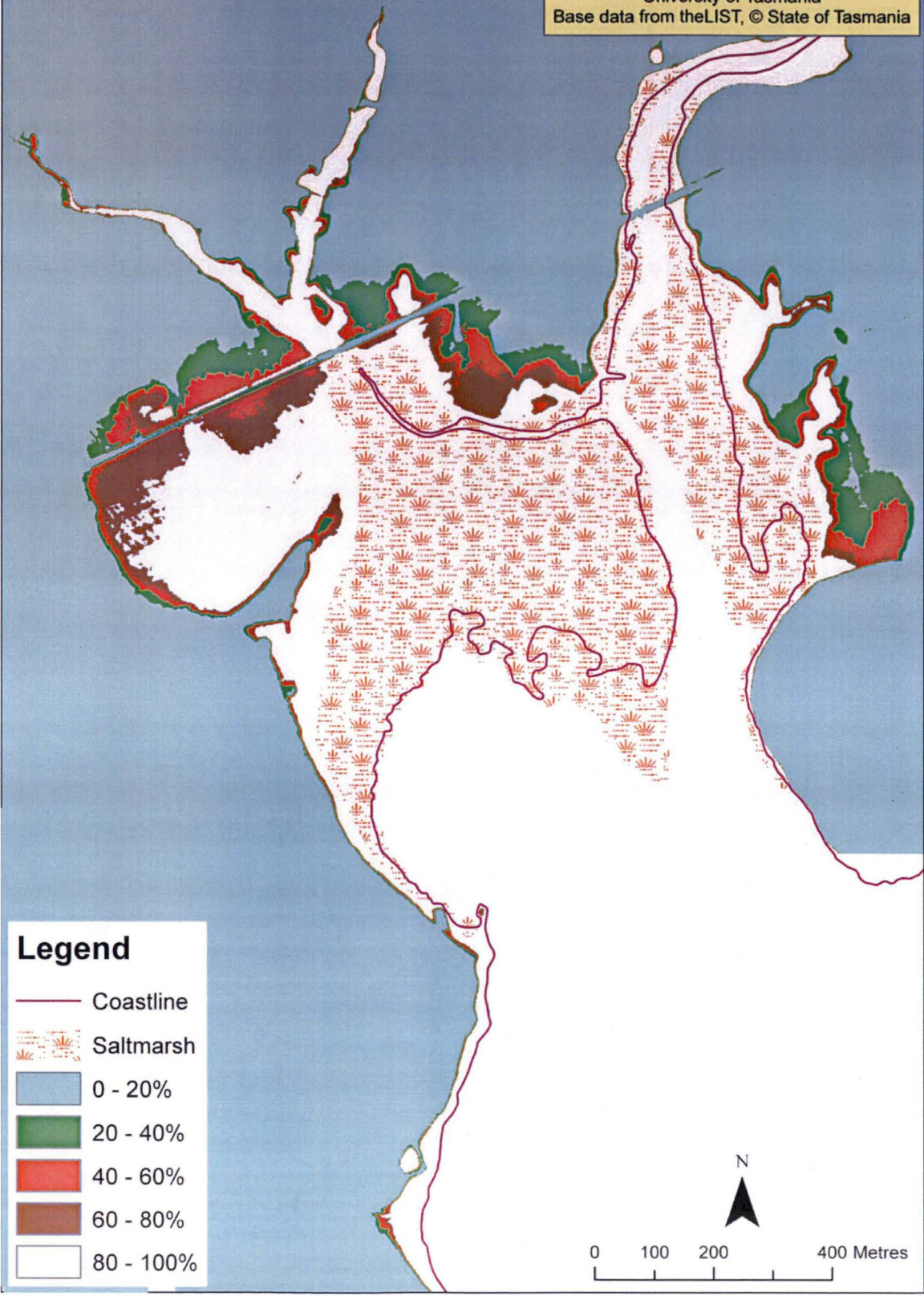


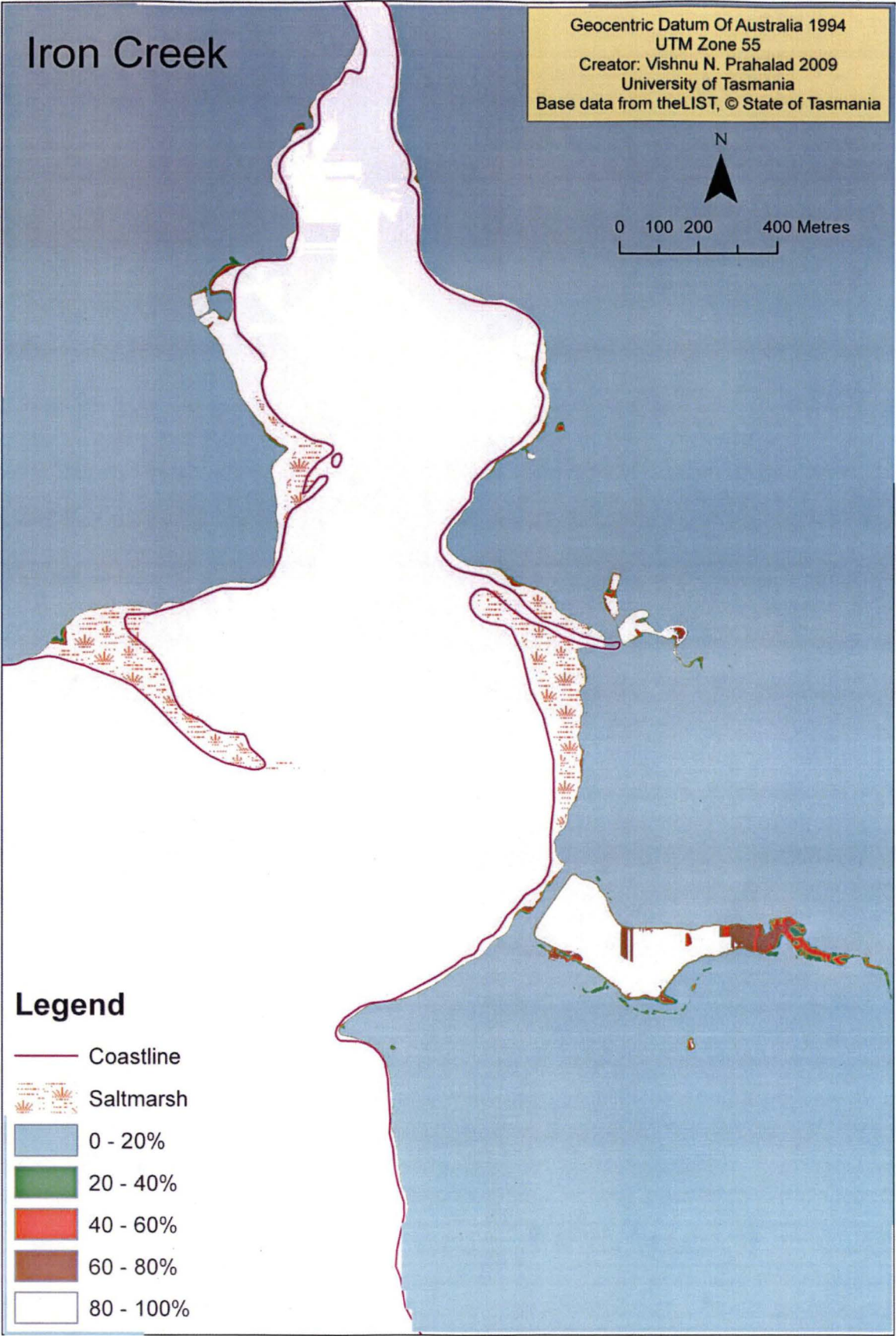




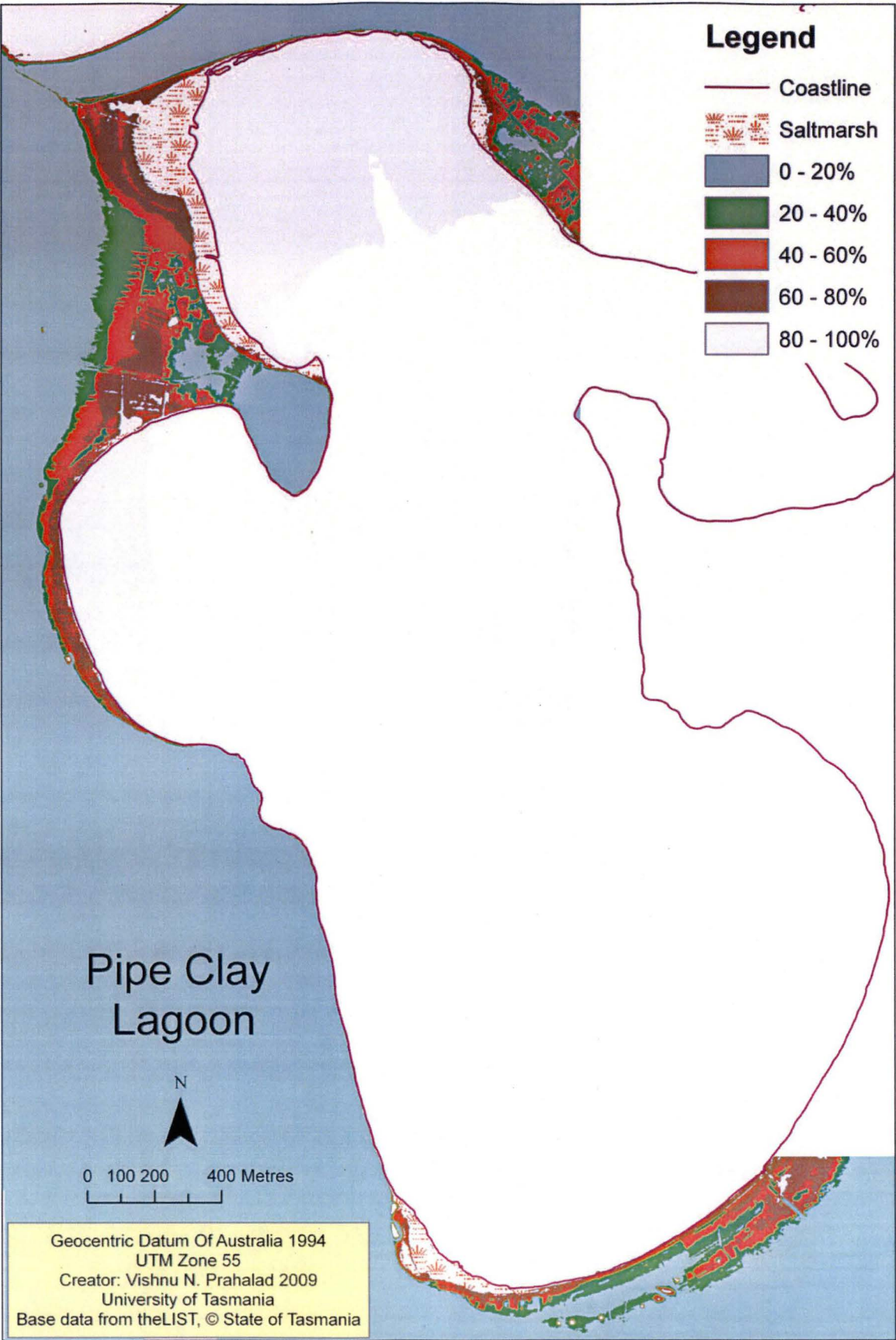
# Orielton Lagoon

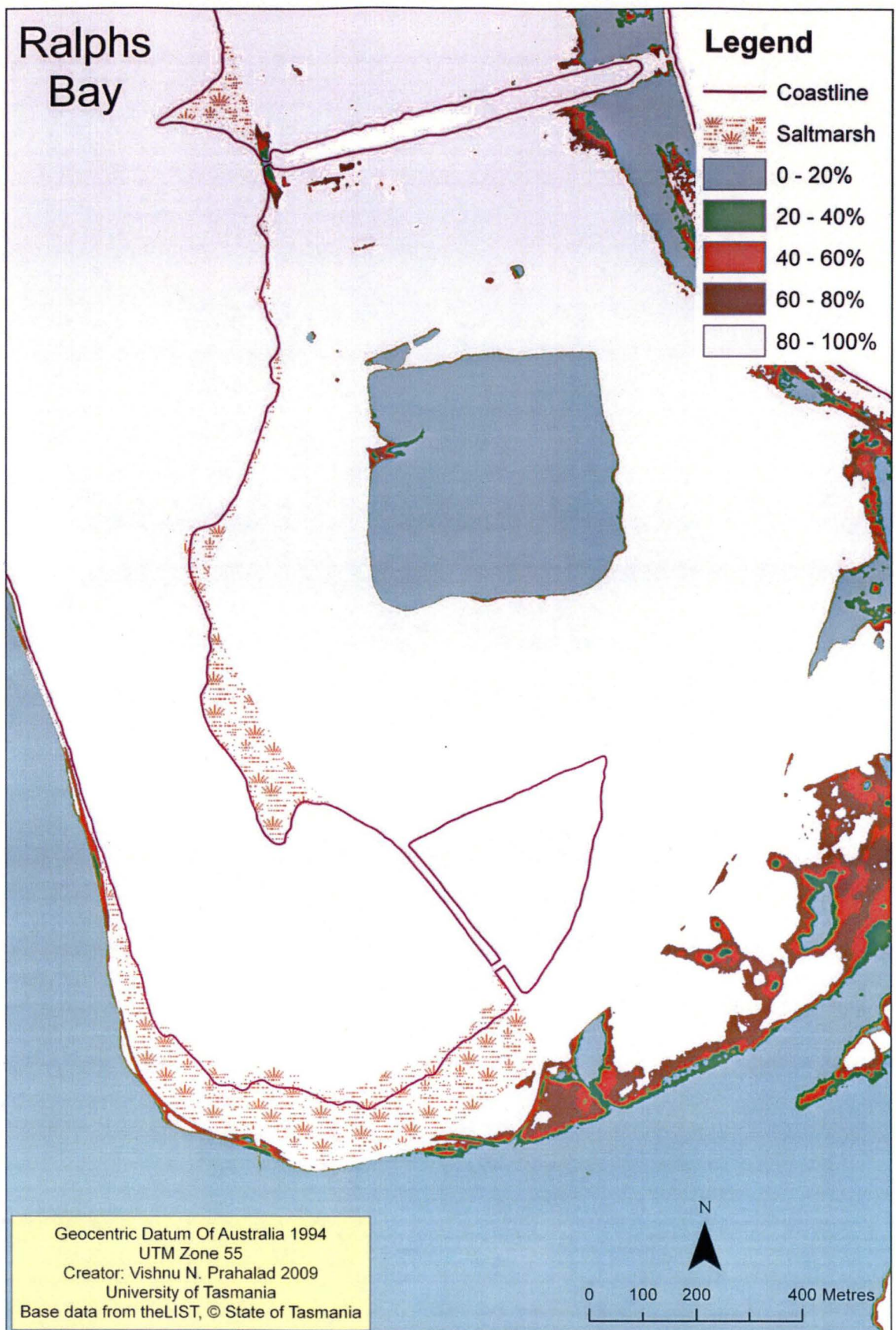
Geocentric Datum Of Australia 1994  
UTM Zone 55  
Creator: Vishnu N. Prahalad 2009  
University of Tasmania  
Base data from theLIST, © State of Tasmania









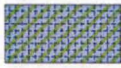


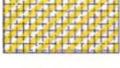




## Appendix III: Vegetation Dominance Groups Maps

Maps of the vegetation dominance groups within each study site are presented here.

### Vegetation Dominance Groups of Saltmarshes 1 - 22

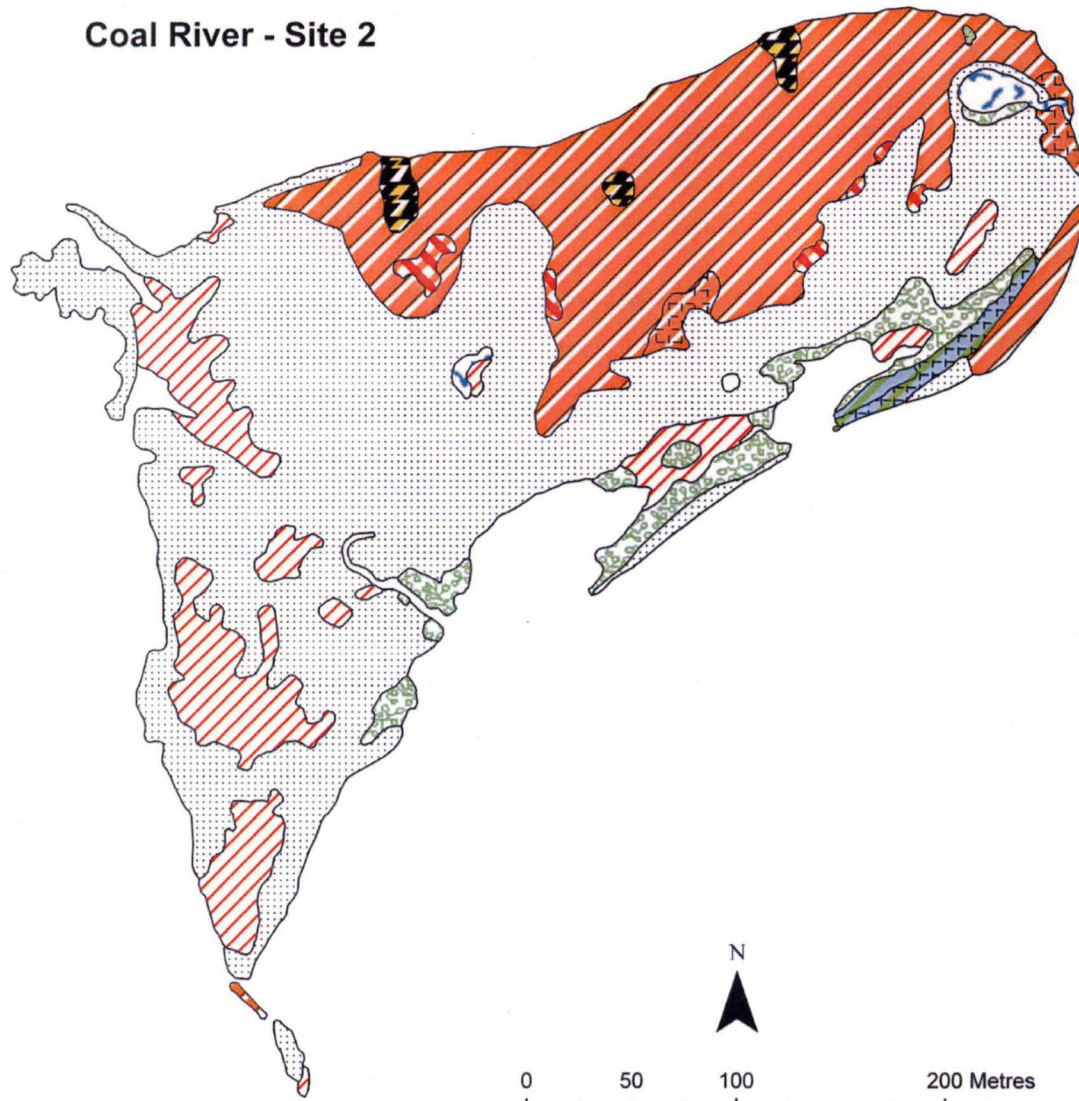
	<i>Juncus kraussii</i>		Not Saltmarsh
	<i>Juncus/Poa/Austrostipa</i>		<i>Poa</i> spp.
	<i>Juncus/Tecticornia</i>		<i>Puccinellia/Sarcocornia</i>
	<i>Juncus/Austrostipa</i>		<i>Suaeda australis</i>
	<i>Ficinia/Juncus</i>		<i>Sarcocornia blackiana</i>
	<i>Distichlis distichophylla</i>		<i>Sarcocornia</i> spp.
	<i>Gahnia/Juncus</i>		<i>Sarcocornia quinqueflora</i>
	<i>Gahnia/Tecticornia</i>		<i>Samolus repens</i>
	<i>Gahnia/Austrostipa</i>		<i>Samolus/Suaeda</i>
	<i>Apodasmia/Poa</i>		<i>Austrostipa stipoides</i>
	<i>Gahnia filum</i>		<i>Stipa/Tecticornia</i>
	<i>Atriplex cinerea</i>		<i>Tecticornia arbuscula</i>
	<i>Atriplex paludosa</i>		<i>Wilsonia backhousei</i>
	<i>Disphyma crassifolium</i>		<i>Wilsonia humilis</i>
	<i>Ficinia nodosa</i>		<i>Disphyma/Sarcocornia</i>
	<i>Hemichroa pentandra</i>		Bare Ground (Salt Pans/Pools)

**Coal River - Site 1**

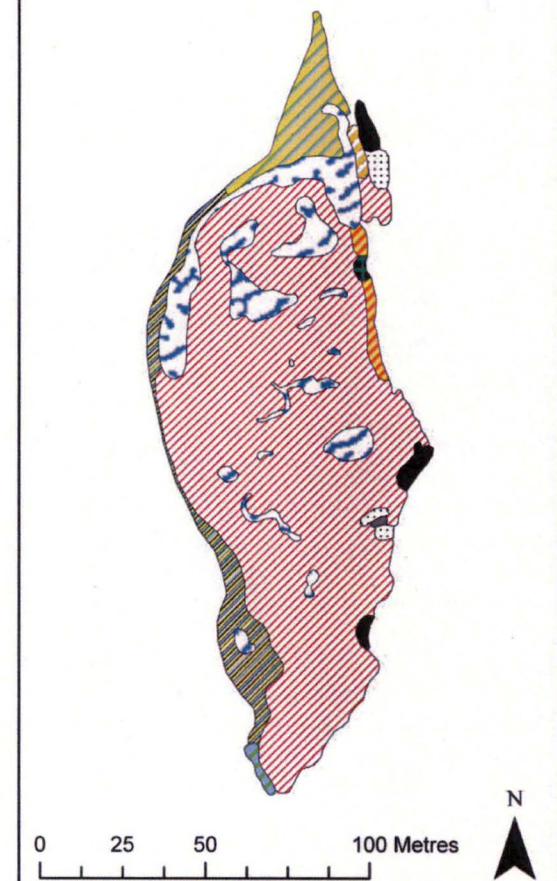




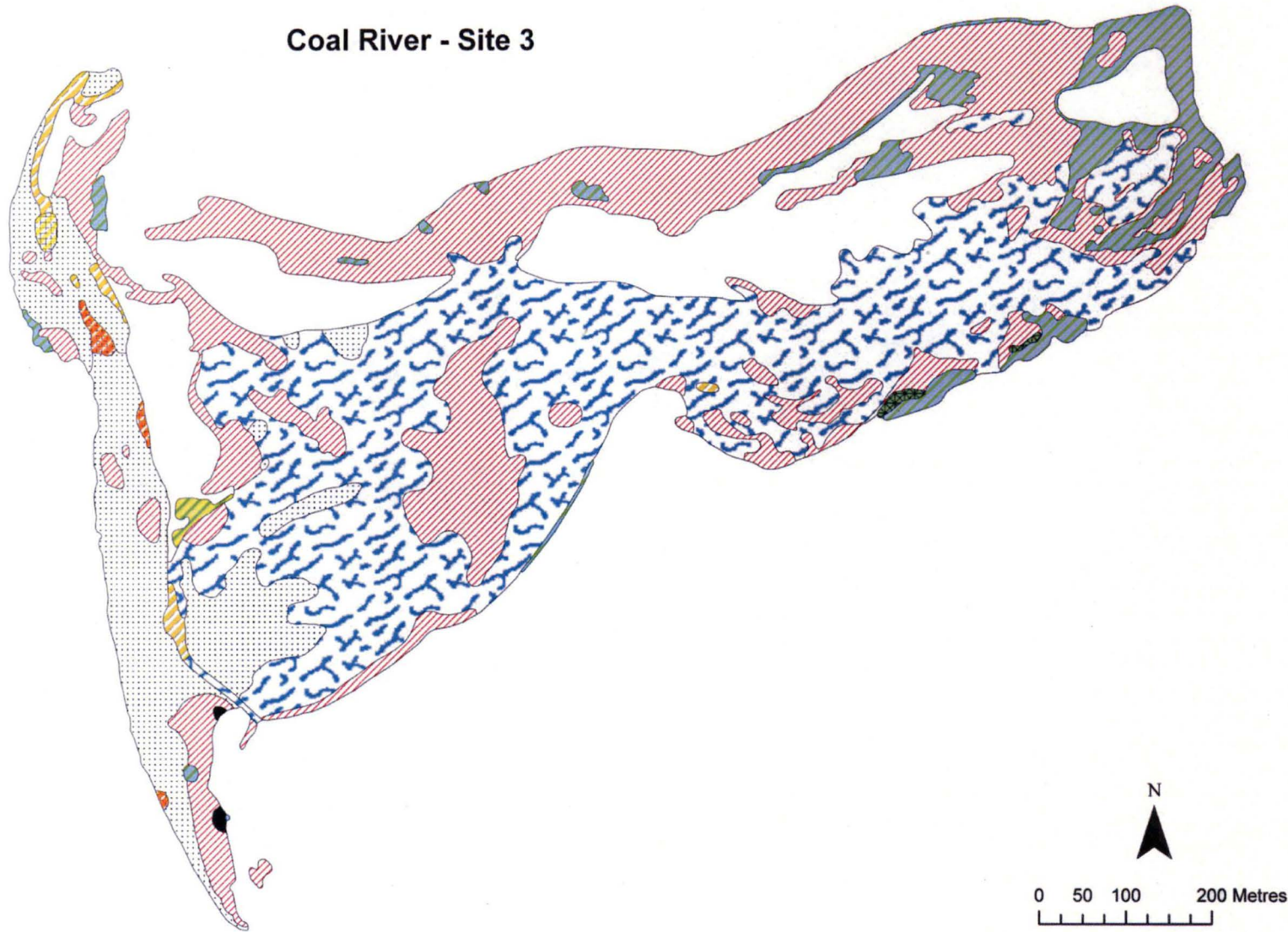
Coal River - Site 2



Coal River - Site 4

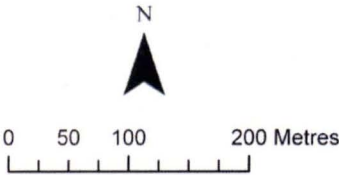
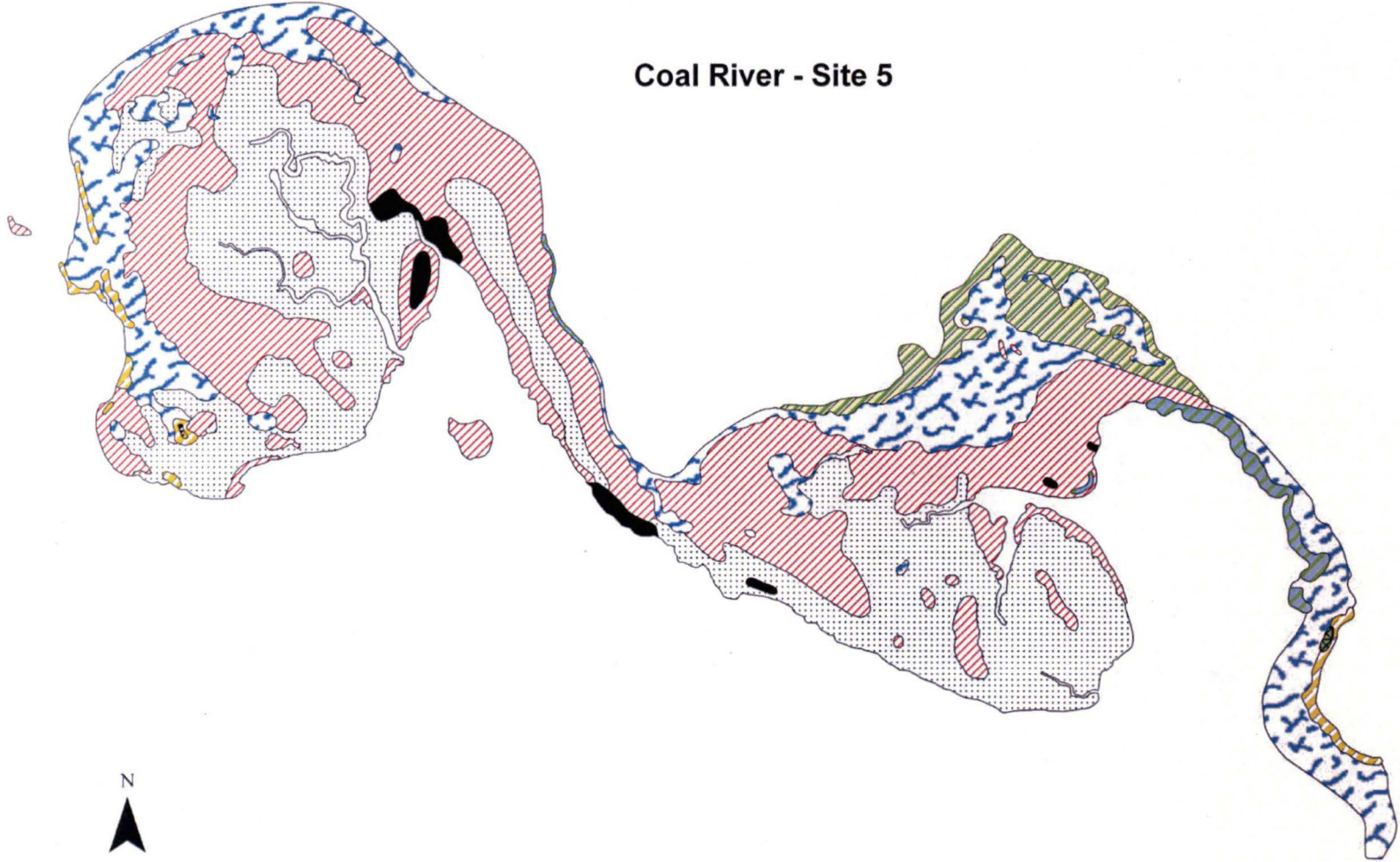


### Coal River - Site 3

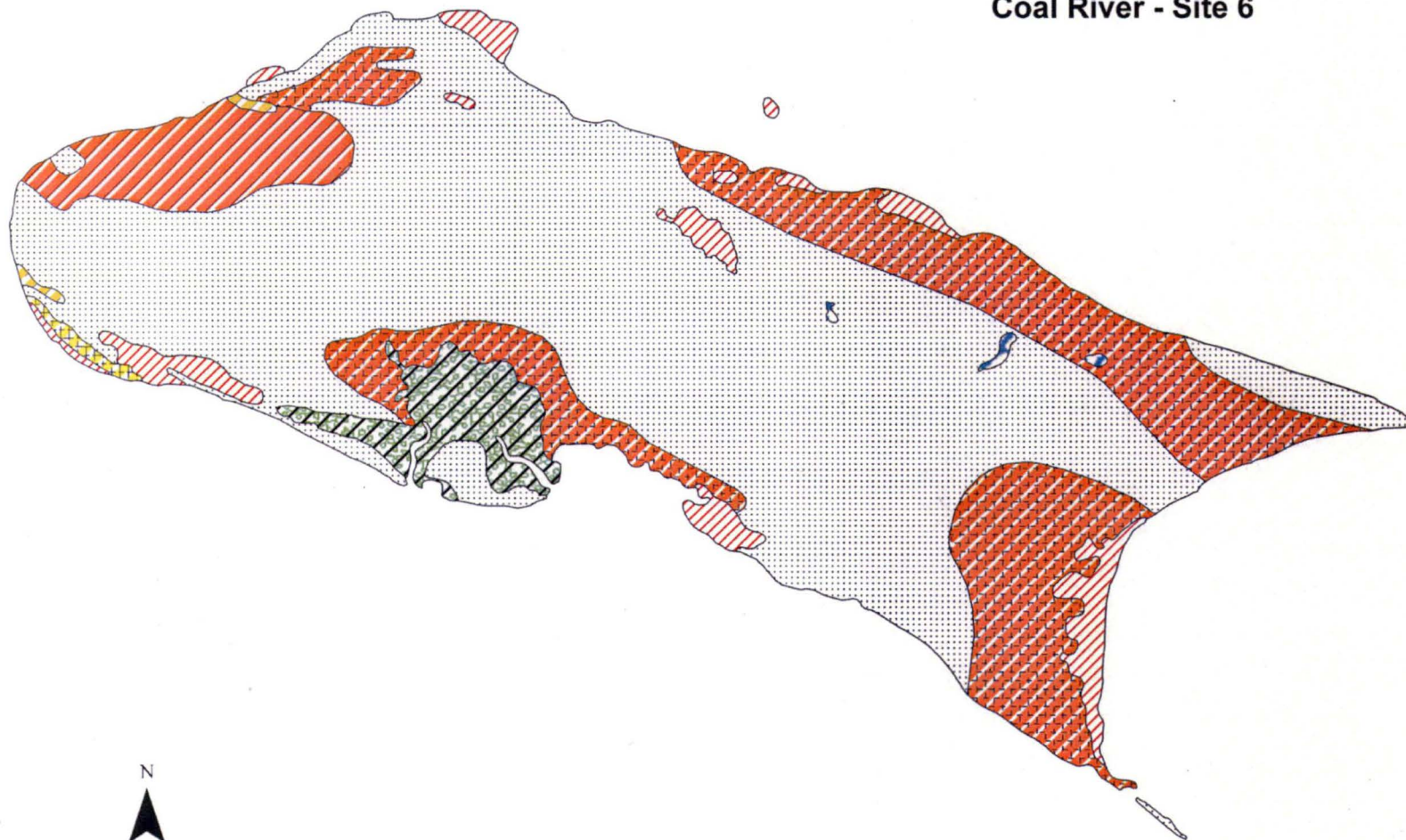




Coal River - Site 5

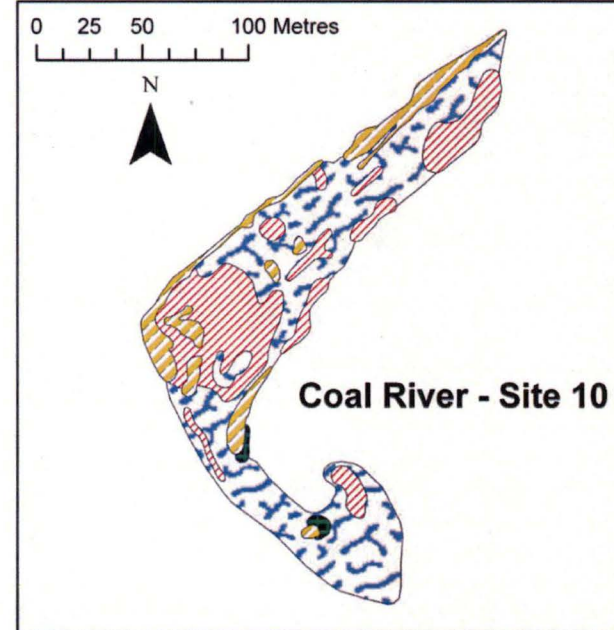
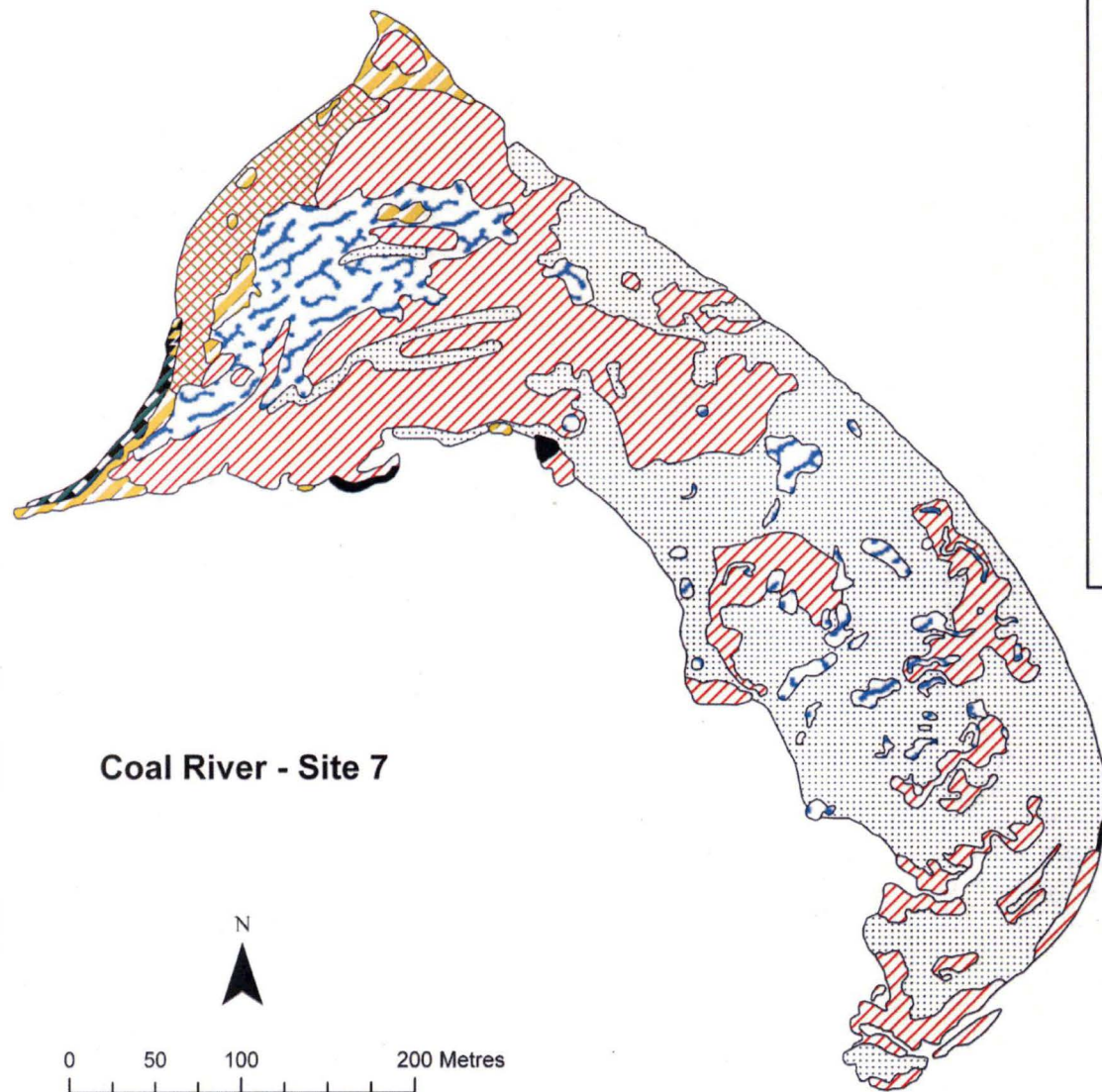


# Coal River - Site 6

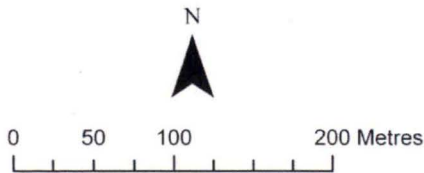
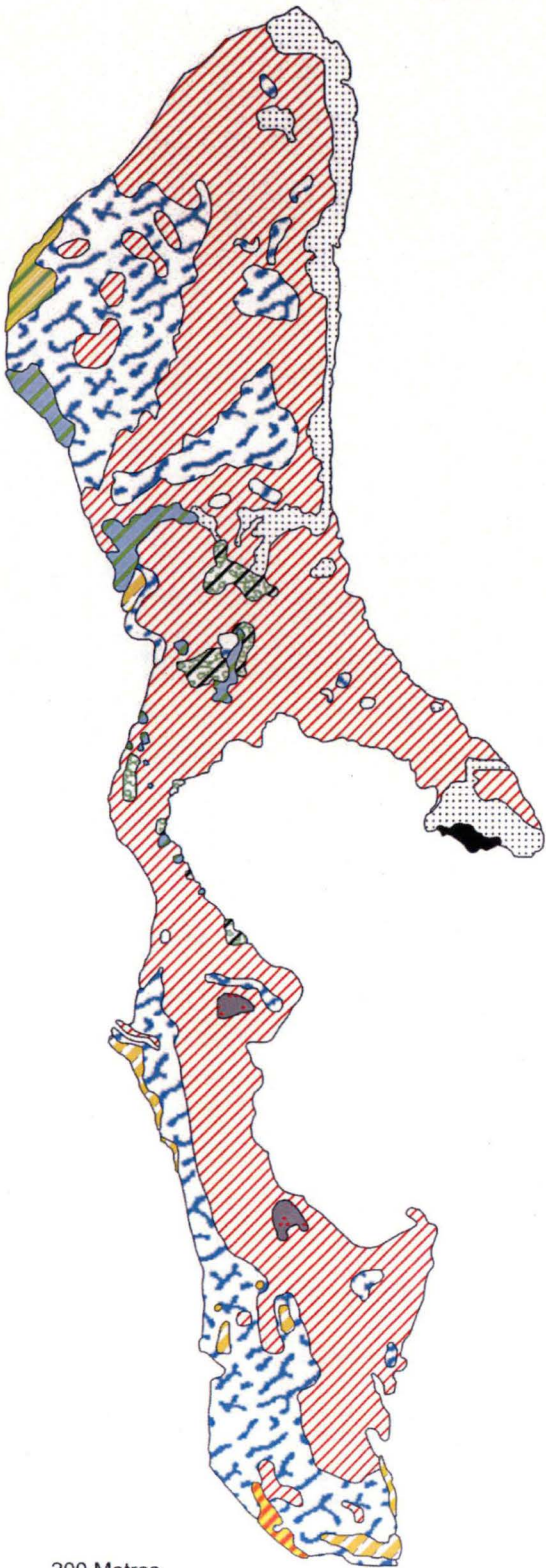


0 50 100 200 Metres





Duckhole Rivulet - Site 8

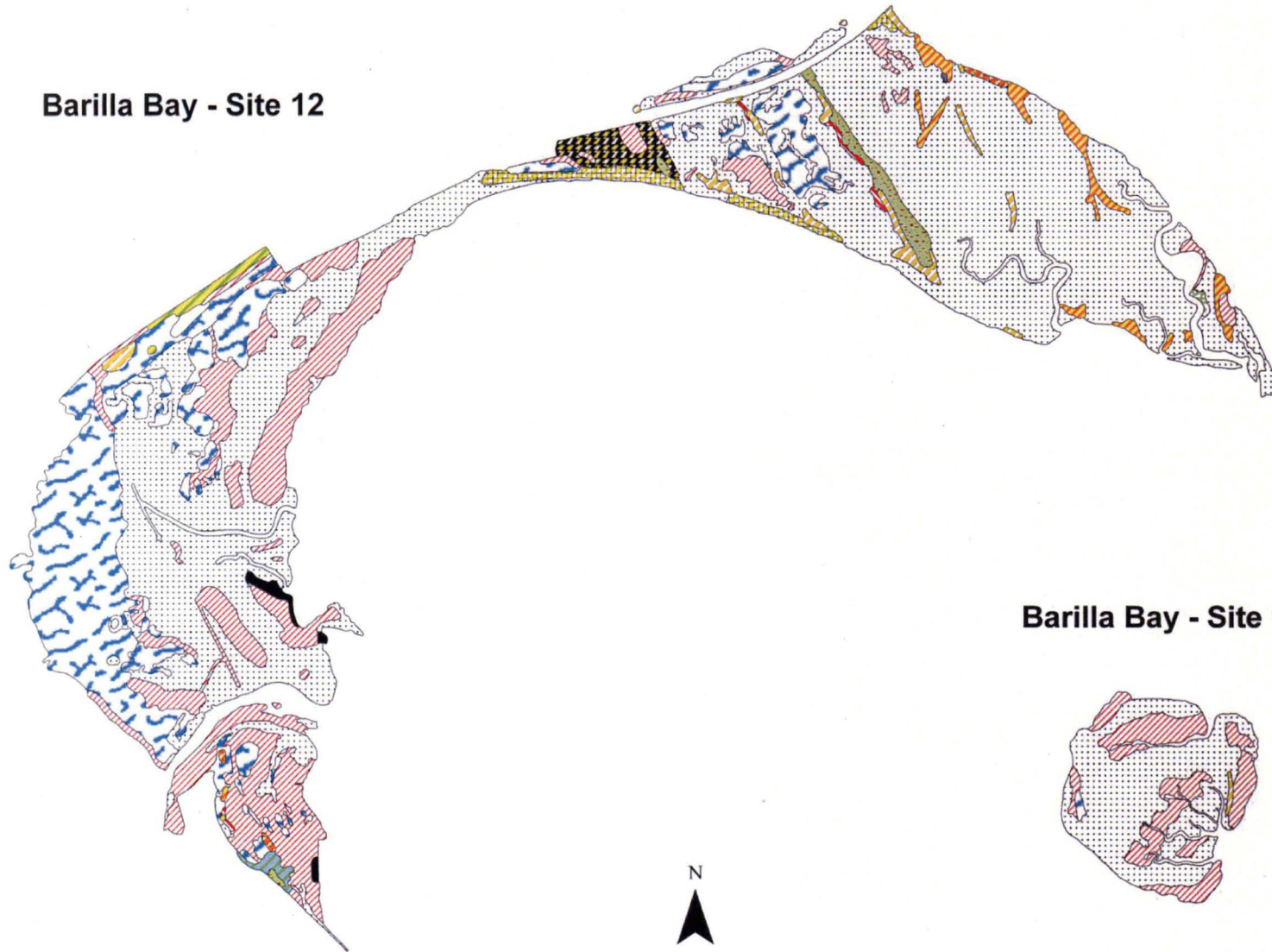




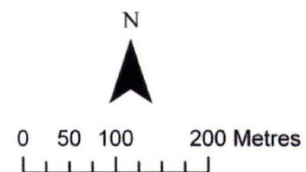
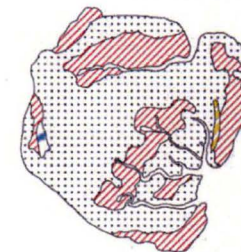
## Duckhole Rivulet - Site 9



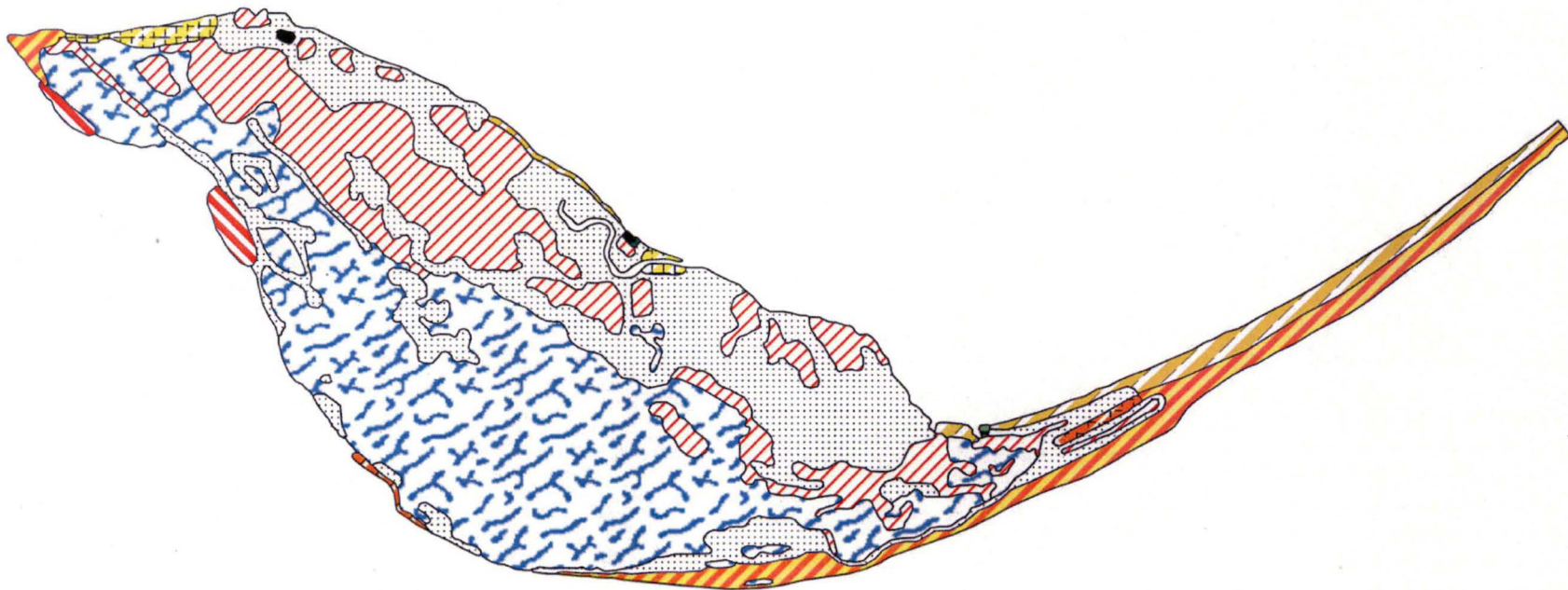
**Barilla Bay - Site 12**



**Barilla Bay - Site 13**



# Barilla Bay - Site 14

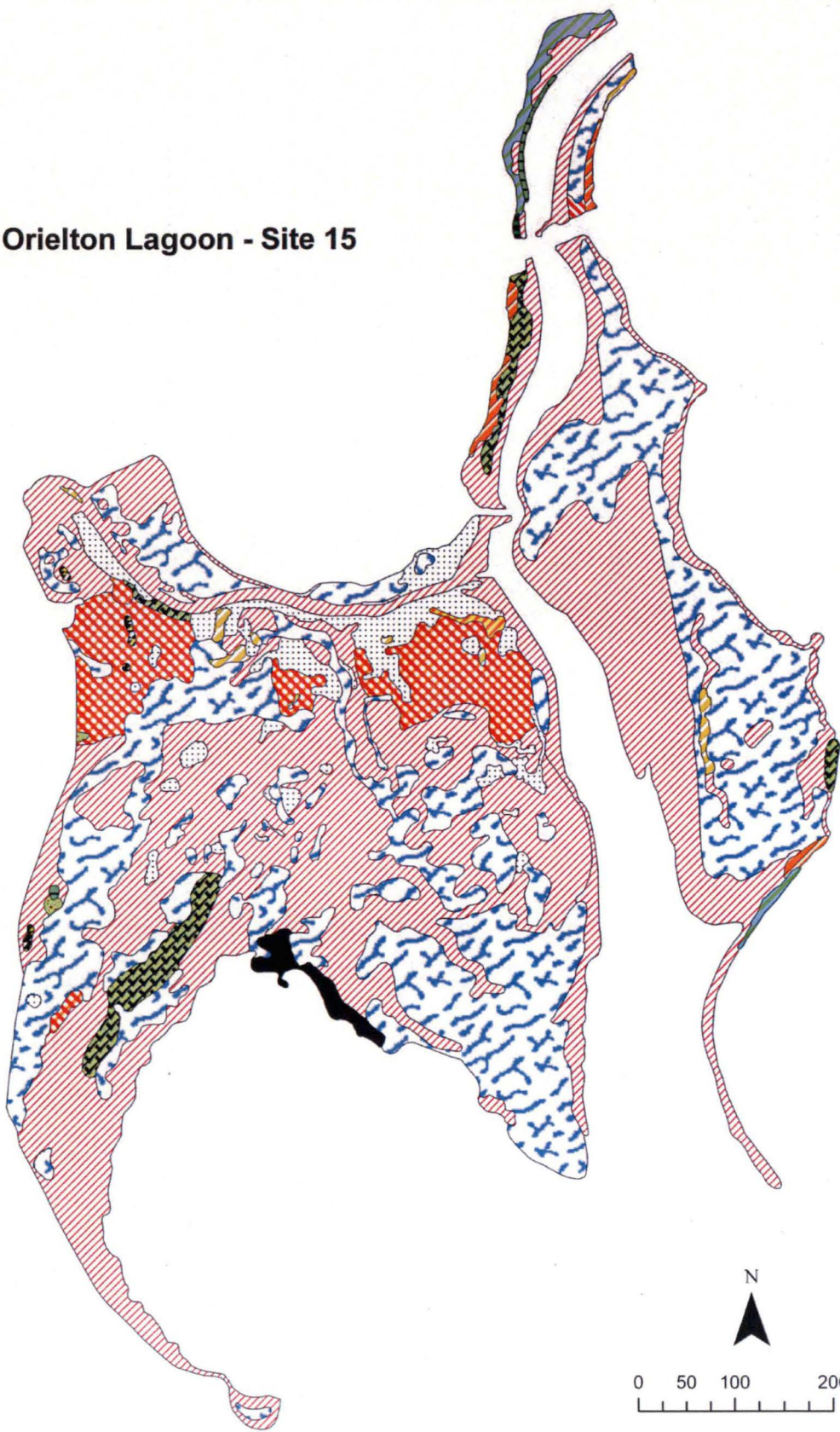


0 50 100 200 Metres

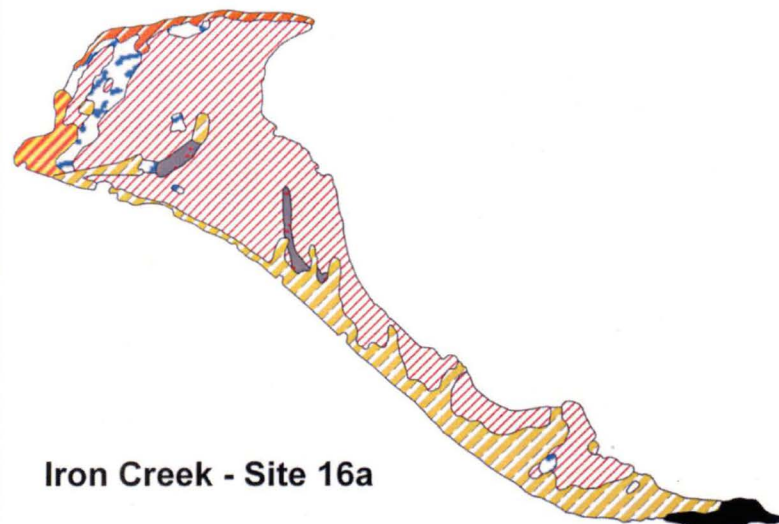




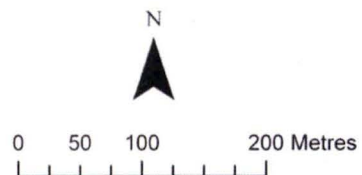
Orielton Lagoon - Site 15



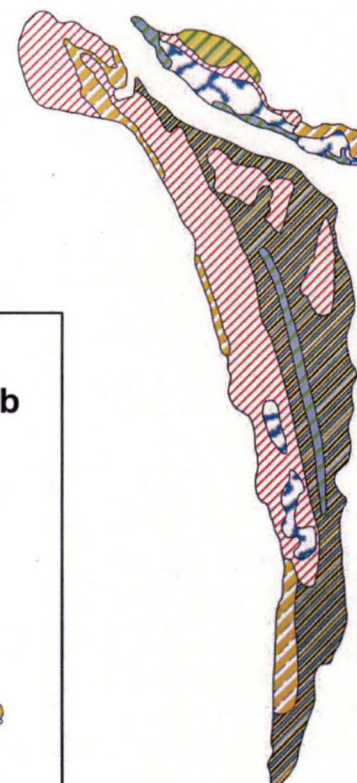




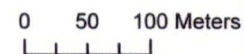
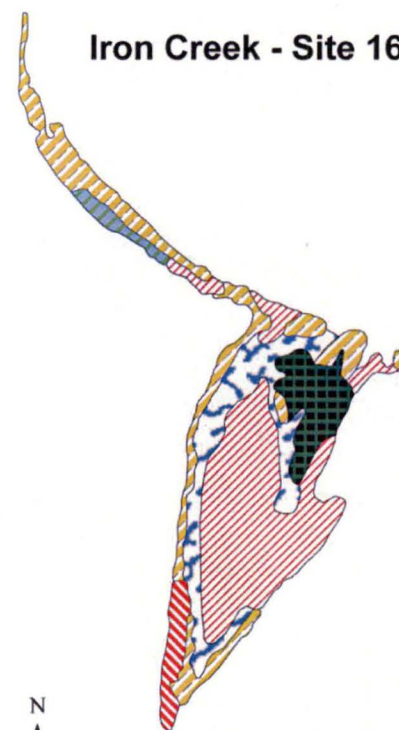
**Iron Creek - Site 16a**

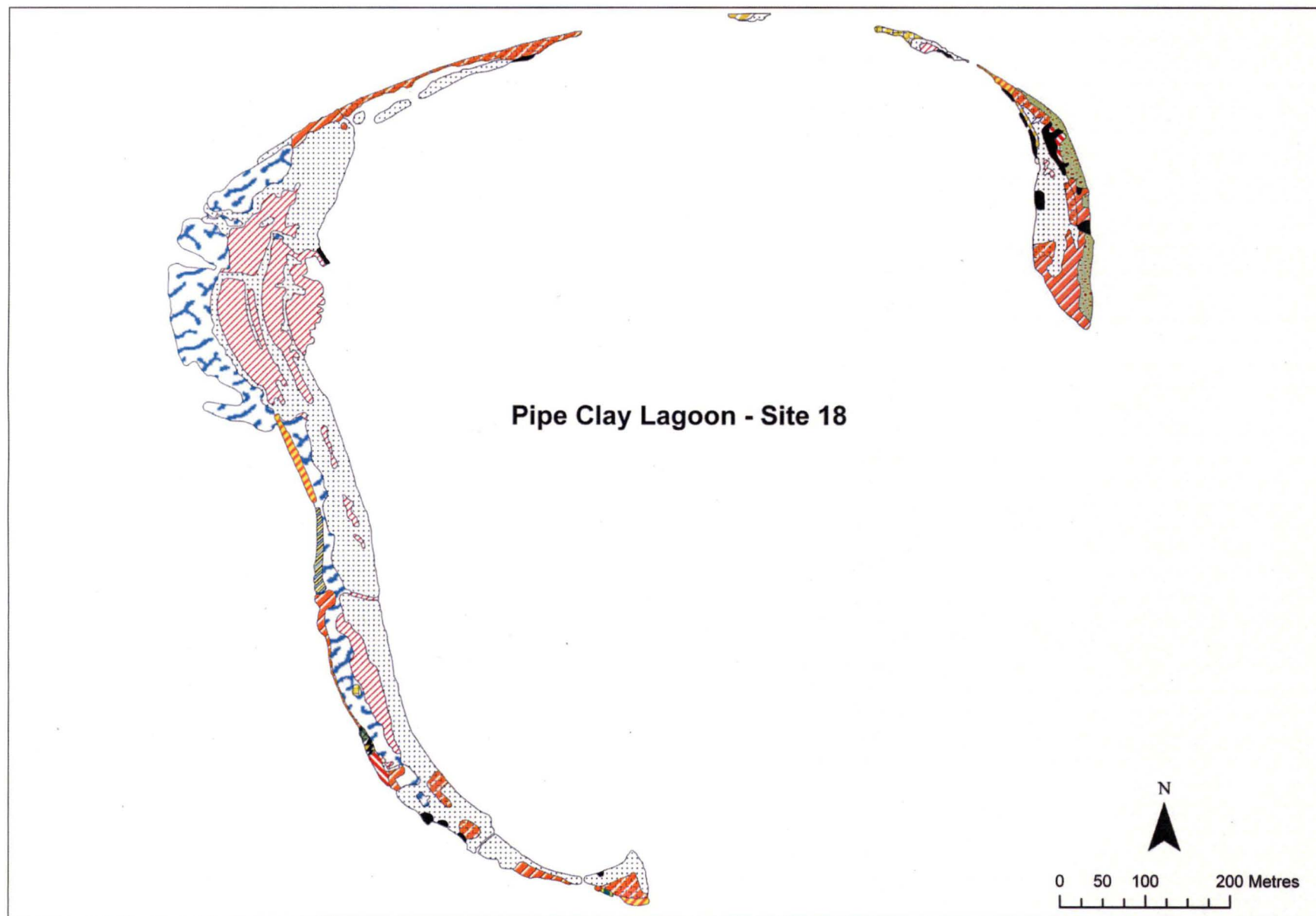


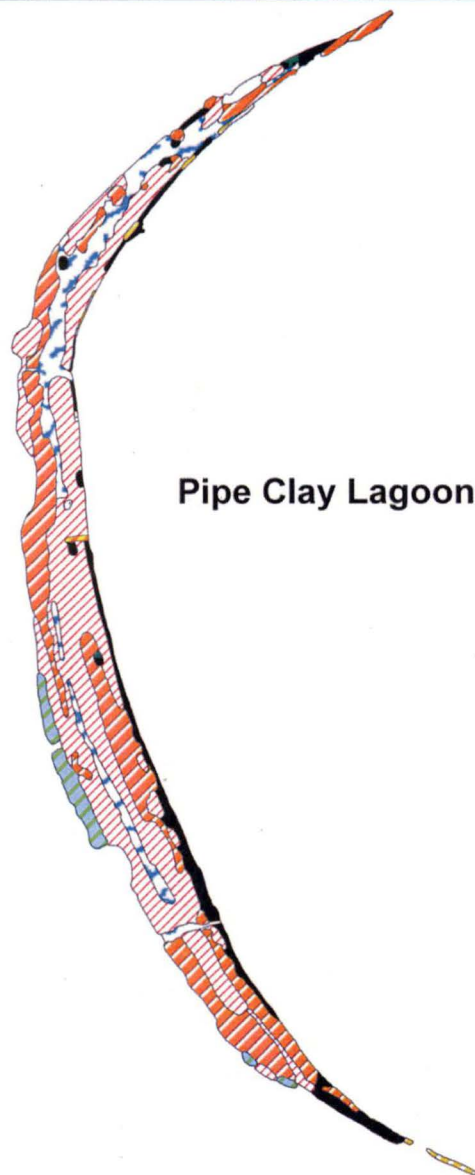
**Iron Creek - Site 17**



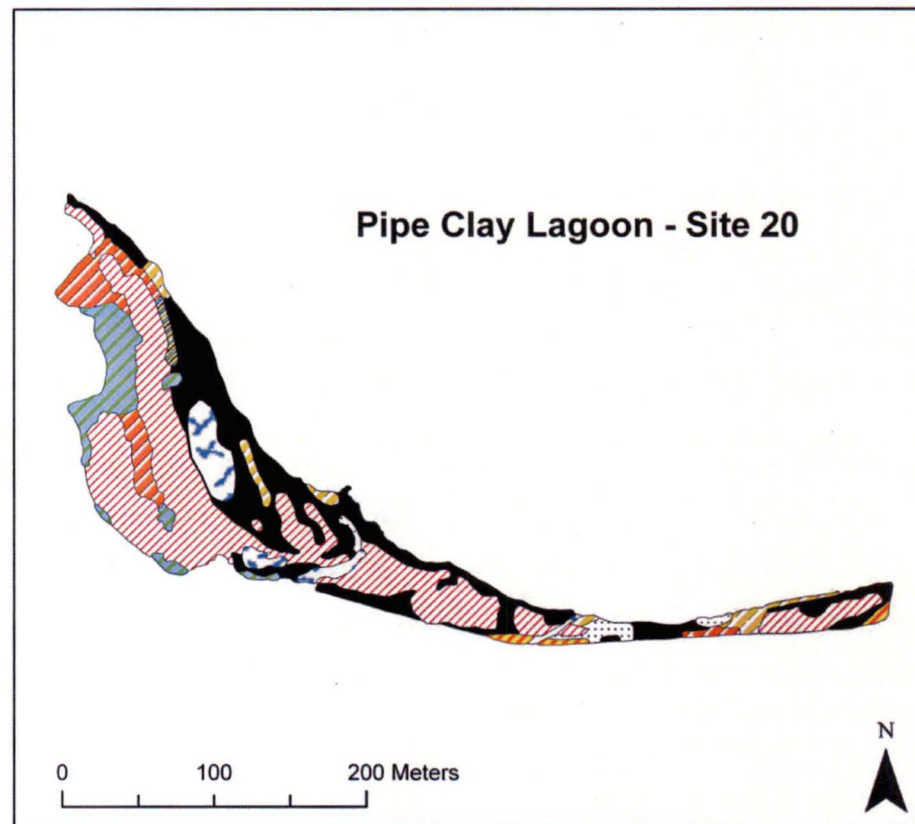
**Iron Creek - Site 16b**



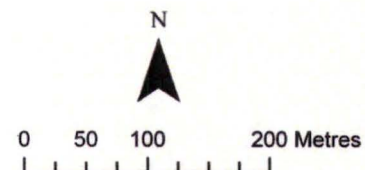




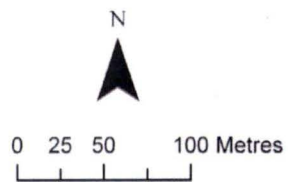
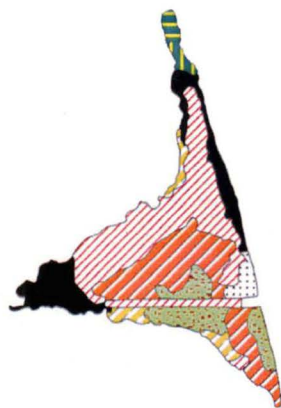
Pipe Clay Lagoon - Site 19



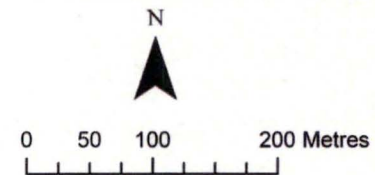
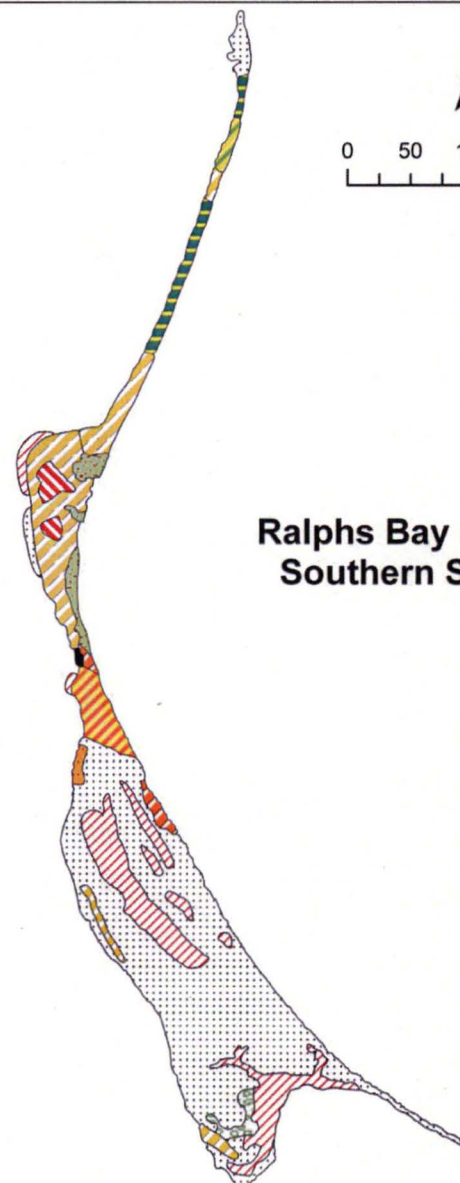
Pipe Clay Lagoon - Site 20



**Ralphs Bay - Site 21  
Northern Section**

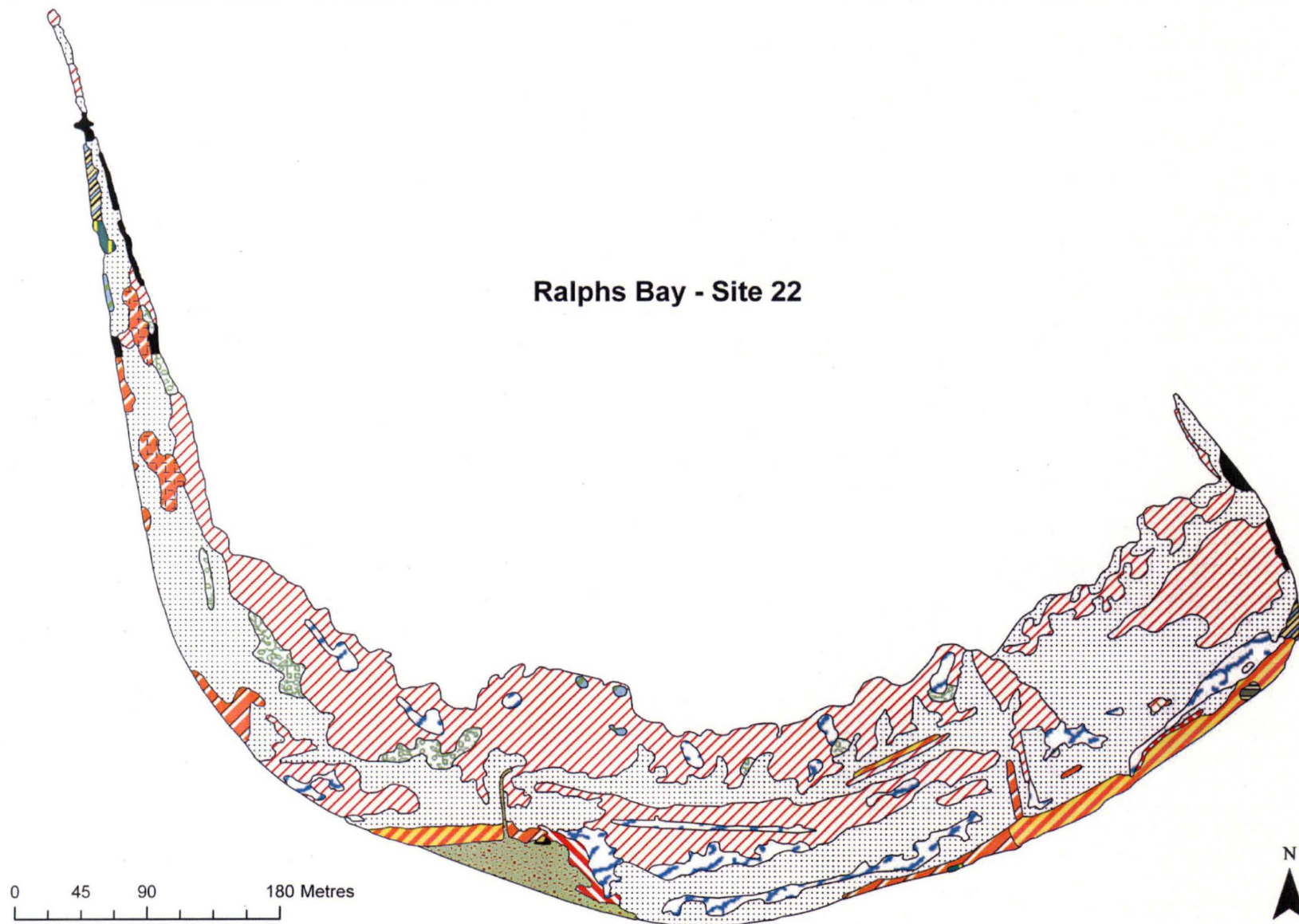


**Ralphs Bay - Site 21  
Southern Section**





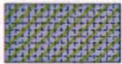





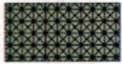



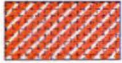






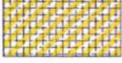

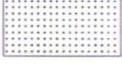







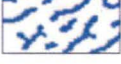




**Ralphs Bay - Site 22**



# Vegetation Dominance Groups of Saltmarshes 1 - 22

	<i>Juncus kraussii</i>		Not Saltmarsh
	<i>Juncus/Poa/Austrostipa</i>		<i>Poa</i> spp.
	<i>Juncus/Tecticornia</i>		<i>Puccinellia/Sarcocornia</i>
	<i>Juncus/Austrostipa</i>		<i>Suaeda australis</i>
	<i>Ficinia/Juncus</i>		<i>Sarcocornia blackiana</i>
	<i>Distichlis distichophylla</i>		<i>Sarcocornia</i> spp.
	<i>Gahnia/Juncus</i>		<i>Sarcocornia quinqueflora</i>
	<i>Gahnia/Tecticornia</i>		<i>Samolus repens</i>
	<i>Gahnia/Austrostipa</i>		<i>Samolus/Suaeda</i>
	<i>Apodasmia/Poa</i>		<i>Austrostipa stipoides</i>
	<i>Gahnia filum</i>		<i>Stipa/Tecticornia</i>
	<i>Atriplex cinerea</i>		<i>Tecticornia arbuscula</i>
	<i>Atriplex paludosa</i>		<i>Wilsonia backhousei</i>
	<i>Disphyma crassifolium</i>		<i>Wilsonia humilis</i>
	<i>Ficinia nodosa</i>		<i>Disphyma/Sarcocornia</i>
	<i>Hemichroa pentandra</i>		Bare Ground (Salt Pans/Pools)

Note: This legend is a part of the thesis titled "Long term changes in south east Tasmanian saltmarshes" by Vishnu N. Prahalad (2009). This page has been left unattached to the thesis on purpose, which is to use it to decode the vegetation maps provided in Appendix III of the thesis. Please place this page just after the last page of the thesis after use (or if found separated from the thesis).